June 5, 1998

Draft Final Report

Non-groundwater Pathways, Human Health and Ecological Risk Analysis for Fossil Fuel Combustion Phase 2 (FFC2)

Prepared for

Office of Solid Waste
401M St., SW (5307W)
Washington, DC 20460

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Prepared by

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Research Triangle Park, NC

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This is a draft document that has not undergone a thorough internal or external review. It does not represent Agency policy.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

EXECUTIVE SUMMARY

Wastes from the combustion of fossil fuels are exempt from regulation as hazardous wastes under the Resource Conservation and Recovery Act (RCRA) pending completion of a Report to Congress by the Environmental Protection Agency (EPA) that details the need for more stringent regulation of these wastes. In 1988 EPA issued a Report to Congress addressing the risks associated with the disposal of wastes generated by the combustion of coal in the electric utility industry. This was followed, in 1993, by a Regulatory Determination that in essence stated that such wastes, as then managed, posed no risks that would justify special regulation. However, this work was limited in its scope. Not only was it limited to the electric power industry, it was also limited to but one fuel (coal), and to but four high-volume wastes when these wastes are managed alone. Certain categories of fossil fuel combustion (FFC) wastes were not permanently excluded from regulation by the 1993 Regulatory Determination pending additional investigation by EPA.

The purpose of this study is to characterize the potential human health and ecological risks associated with the management and use of the remaining wastes in support of a Report to Congress for these wastes. To do this, EPA performed a multi-pathway human health and ecological risk analysis. Both direct and indirect exposure pathways were assessed for both human and ecological receptors. A deterministic approach was used to estimate point values for individual risks to human health and, by inference, population- and community-level risks to wildlife in generalized aquatic and terrestrial ecosystems.

The universe of wastes considered in this analysis was divided into four sectors: (1) coal-fired utility co-managed wastes, (2) oil-fired utility wastes, (3) fluidized bed combustion wastes, and (4) non-utility FFC wastes. EPA found that each sector represented a distinct generator population with distinct byproduct characteristics and, therefore, elected to study the risks from each sector separately. Only waste management or disposal practices that are currently used were assessed and included management in surface impoundments, management in landfills, beneficial use as an agricultural soil amendment, and beneficial use as minefill.

Given the number of facilities within each industry sector and the geographic distribution of facilities, site specific analyses were not possible for any of the industry sectors. Therefore, hypothetical wastes and waste management units (WMU's) were developed for each industry sector based on available data. Individual facilities for all industry sectors are distributed nationally with the exception of oil-fired utilities and, therefore, data on waste matrices and onsite waste management practices were taken from sector-specific reports (e.g., comanaged coal ash) and national data compilations (e.g., industrial boilers). National data were also used to characterize the environmental setting for each exposure scenario for human and ecological receptors. For example, meteorological data and soil characteristics were identified from national data distributions. Previous analyses conducted by OSW were used to determine appropriate human health and ecological exposure factors (e.g., EPA's Exposure Factors Handbooks) as well as receptor distances from waste management units. Because the analysis was not site-specific, ecological receptors were presumed to co-occur with waste management units.

The types of contaminant release and transport that were assessed included: leaching to groundwater; soil erosion and runoff; and particulate air emissions and dispersion. For both the human health and ecological risk assessment, exposures were assessed for direct contact with contaminated media as well as ingestion of contaminated food items or prey. Direct pathway exposures stem from ingestion of contaminated groundwater and soil as well as direct inhalation of contaminated air. Indirect pathway exposures stem from ingestion of food items (e.g., plants, beef, or dairy products) that have been grown in, or raised on, areas that have been contaminated from offsite migration of constituents from the waste or direct application of the waste material as agricultural soil amendment.

Several different types of receptors were considered in this analysis in order to represent the full spectrum of behaviors associated with “typical” exposures (e.g., adult residents) and high exposures indicative of certain subpopulations (e.g., farmers). Including the highly exposed subpopulations ensures that significant subpopulations of potentially exposed humans and wildlife are protected. For example, potentially more exposed human populations are identified based on behaviors and activities (e.g., consumption of self-caught fish or growing foods for home consumption), or based on their proximity to a contaminant source. Selected receptors include: (1) adult resident, (2) home gardener, (3) farmer, (4) child of farmer, and (5) fisher. This array of receptors covers all plausible exposure pathways. The child receptor was included to represent a sensitive subpopulation potentially exposed to all of the most plausible direct and indirect exposure pathways. With respect to ecological risks, receptors are chosen to represent the full trophic continuum of wildlife species populations and communities that are essential to the structure and function of ecosystems.

In the initial stages of this analysis, a bounding assessment was performed. For the human health risk assessment, the bounding analysis estimated risk for the most exposed individual living in the proximity of the waste management unit (i.e., off-site exposure). For the ecological risk assessment, the bounding analysis estimated risks to wildlife living in the vicinity of units that were characterized by conservative management and use practices (e.g., maximum waste constituent concentrations; maximum unit sizes). For this deterministic assessment, all parameters were set at high-end values (5^{th} or 95^{th} percentile depending on the parameter’s correlation with risk) and risks were estimated. In evaluating potential health and ecological risks, the bounding analysis was designed with very conservative assumptions intended capture the tail of the risk distribution (i.e., > 95^{th} percentile). Constituents, waste streams, or waste management units/disposal practices that showed no excess risk in the bounding analysis were excluded from further analysis.[1] The results of the bounding analysis indicated the need for further refinements in the form of high-end analyses.

For the high-end analysis, both high-end and central tendency risk estimations are presented. High-end analyses are intended to represent risk above the 90th percentile of the distribution of individual risk in a population but not higher than the individual with the highest risk. Central tendency estimates are intended to represent risks near the 50th percentile of the distribution of individual risk or, in other words, the risk to someone with a more typical exposure.

For the central tendency risk estimation, all parameters are set to their central tendency value (usually the median value of the distribution) and the risk or hazard quotient for each constituent is calculated. The high-end risk estimation utilizes a two parameter high-end assessment methodology. With this approach, two driving risk parameters at a time are set to their high-end value while the remainder of the parameters are set at central tendency and risk values are calculated accordingly. This continues until risk values are calculated for all possible high-end parameter combinations. As suggested under the description of the bounding analysis, high-end and central tendency estimates of ecological risk refer only to the management and use parameters; the ecotoxicological data and exposure assumptions remained constant throughout all tiers of the ecological risk assessment.

In analyses such as this, uncertainty is introduced in a number of ways. In general, the major sources of uncertainty are parameter uncertainty and model uncertainty. Model uncertainty is associated with all models used in all phases of a risk assessment. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in models due either to increased complexity or to a lack of data on that variable. Other than these inherent model uncertainties, there were two major sources of uncertainty in this analysis. These included waste characterization uncertainty and waste management practice uncertainty. Many of the sources used for the waste characterization and waste management unit parameter development had limited data or were based on small sample sizes. This causes problems when trying to infer from a limited data set to the population of wastes or waste management units and practices. In addition, the variability in many of the input parameters introduces significant uncertainty in the risk results, particularly since data were frequently selected from national (versus regional or site-based) data distributions.

All of these uncertainties need to be taken into account when attempting to characterize the risks from the management and use of fossil fuel wastes. The human health risk results

[1] Throughout this assessment, excess risk was defined for humans as risk values greater than or equal to 1E-6 for carcinogens or hazard quotients greater than or equal to 1.0 for non-carcinogens. For ecological receptors, excess risk was defined as hazard quotients greater than 1.0.

indicate that arsenic risks from ingestion are of concern for some of the waste management unit/receptor combinations and that chromium VI risks from inhalation are of potential concern. Barium and thallium also had hazard quotients that indicated that these constituents were of potential concern. The driving risk parameter was starting waste concentration. Other important parameters include area of the waste management unit, exposure duration, and distance to receptor. The ecological risk results indicate that amphibians and wildlife that utilize surface impoundments as part of their natural habitat may be at risk from several metals, particularly selenium and aluminum. Although elevated risks were also shown for mercury, the paucity of sampling data makes the risks results difficult to interpret.

Consequently, there are several uncertainty issues that should be considered when evaluating the results for this analysis. Many of the assumptions required in this analysis are, as a matter of policy or past practice, conservative and intended to be protective. For some environmental settings and waste management units, these assumptions likely lead to an overstatement of risks. However, EPA has identified the major uncertainty issue in this assessment: the waste characterization. Modeling constituent concentrations that reflect current comanaged FFC residuals is especially important since the waste concentration tends to be the driving risk parameter. The representativeness and statistical rigor in the sampling data used in the waste characterization have important implications for the risk results and subsequent decisions supported with this analysis.

For the draft analysis, uncertainty has been addressed qualitatively. Given the high degree of variability for many of the driving risk parameters, and the number of data deficiencies inherent in a national-scale (versus site-based) analysis, a quantitative uncertainty analysis will be performed to ascertain how these uncertainties influence risk results.

1.0 BACKGROUND, PURPOSE, AND SCOPE

1.1 Background

Wastes from the combustion of fossil fuels are exempt from regulation as hazardous wastes under the Resource Conservation and Recovery Act (RCRA) pending completion of a Report to Congress by the Environmental Protection Agency (EPA) that details the need for more stringent regulation of these wastes. In 1988 EPA issued a Report to Congress addressing the risks associated with the disposal of wastes generated by the combustion of coal in the electric utility industry. This was followed, in 1993, by a Regulatory Determination that in essence stated that such wastes, as then managed, posed no risks that would justify special regulation. However, this work was limited in its scope. Not only was it limited to the electric power industry, it was also limited to but one fuel (coal), and to but four wastes (fly ash, bottom ash, boiler slag, and flue gas desulphurization [FGD] sludge) when these wastes are managed alone. Certain categories of fossil fuel combustion (FFC) wastes were not permanently excluded from regulation by the 1993 Regulatory Determination pending additional investigation by EPA. These “remaining wastes” include coal-fired electric utility wastes that are co-managed with other low volume wastes; wastes from coal-fired non-utility industries; fluidized bed combustion wastes and; wastes from the combustion of other fossil fuels.

1.2 Purpose of Study

The purpose of this study is to characterize the potential human health and ecological risks associated with the management and use of the remaining wastes in support of the Report to Congress for these wastes. The risk assessment for the remaining FFC wastes includes: human health risks from groundwater contamination; human health risks from multiple above-ground exposure routes; and ecological risks. This report presents the technical approach to, and results of, the above-ground multipathway human health risk assessment and the ecological risk assessment. The groundwater pathway human health risk assessment, conducted in close coordination with the above-ground multipathway human health risk assessment, is presented in the *Technical Background Document for the Supplemental Report to Congress on Remaining Fossil Fuel Combustion Wastes; Ground-Water Pathway Human Health Risk Assessment*; *Draft Final* (SAIC, 1998a), henceforth referred to as the SAIC Background Document.

The Report to Congress, for which this analysis is being performed, has a legislatively mandated delivery date of September 30, 1998. Following review of the Report to Congress, a regulatory determination for these remaining FFC wastes is scheduled for April 1, 1999. This analysis will be used in concert with other information (i.e., the groundwater risk assessment, damage cases, uncertainty analyses, assessment of coverage by other regulatory programs) to make the Regulatory Determination of whether or not these wastes warrant listing as a hazardous waste under RCRA.

1.3 Scope of Report

The remainder of the report begins by developing the analytical framework for the analysis. Following this, a general overview of the risk assessment methodology that was employed for this assessment is reviewed. Section 3.0 presents the waste stream and waste management unit (WMU) characterizations that were developed to represent the remaining FFC waste universe. Section 4.0 discusses the fate and transport modeling that was performed. A discussion of the human health risk assessment, including a discussion of risk results, follows in Section 5.0 and the ecological risk assessment is discussed in Section 6.0. Uncertainty issues for this analysis are discussed in Section 7.0. These uncertainty issues are combined with the risk results to draw some general conclusions in Section 8.0.

2.0 ANALYTICAL FRAMEWORK AND METHODOLOGY

EPA sought to assess risks from the remaining universe of FFC wastes by utilizing a multi-pathway human health and ecological risk analysis. Both direct and indirect human exposure pathways were assessed as well as ecological exposures. Direct pathway exposures stem from ingestion of contaminated groundwater and soil as well as direct inhalation of contaminated air. Indirect pathway exposures stem from ingestion of plants, beef, or dairy products that have been grown in, or raised on, soil that has been contaminated from offsite migration of constituents from the waste. For this analysis, all constituents are modeled for all pathways for a number of receptors.

2.1 Analytical Framework

The remaining waste universe was segregated into four sectors: coal-fired utility co-managed wastes, oil-fired utility wastes, fluidized bed combustion wastes, and non-utility FFC wastes. EPA found that each sector represented a distinct generator population with distinct byproduct characteristics and therefore, EPA elected to study the risks from each remaining category separately.

All waste stream and waste management unit characterizations for each of the four industry sectors were supplied by to RTI by EPA. For an in depth analysis of these industries, including waste stream and WMU characterization, see the SAIC Background Document. Physical and chemical properties for the constituents of concern are presented in Appendix A. Waste stream concentration characterization and surface impoundment characterization (used in the ecological risk assessment) are discussed in Appendix B.

Given the number of facilities within each industry sector and the geographic distribution of facilities, site specific analyses were not possible for any of the industry sectors. Figures 2-1 through 2-4 show the locations, by state, of facilities for each industry sector. The numbers for the respective states represents the number of facilities within that state. Review of Figures 2-1 through 2-4 gives insight into the geographic distribution of facilities for each industry sector. Individual facilities for all industry sectors are well distributed nationally with the exception of oil-fired utilities. For this reason, national data are assumed protective in all cases and were used to characterize parameters other than waste characterization and onsite WMU's - namely meteorological data, soil parameters, distance to receptor and nearest waterbody, national exposure factors, and offsite WMU's. While it is noted that the geographic distribution of oil-fired utilities is predominately in the northeast and southeast, it is assumed that high-end parameters based on national data would maintain conservatism in the analysis.

Central tendency and high-end meteorological locations were selected based on a sensitivity analysis. Central tendency and high-end values generally will be a median or mean value and a 90th percentile value; however, it is difficult to identify true central tendency and high-end meteorological locations because of the numerous variables associated with each location. Different high-end meteorological parameters will necessarily drive exposures from different pathways. For the non-groundwater pathways analysis, it was decided that the high-end meteorological location would be selected to maximize air transport of contaminants versus other transport mechanisms.

Three sources of contamination from air transport were assessed -- particulate air concentration, wet particle deposition, and dry particle deposition. Wet deposition of particles occurs during precipitation and therefore is maximized in areas with high rainfall. However, rainfall also reduces emissions and therefore air concentrations and dry deposition. Because dry particle deposition predominates over wet particle deposition with respect to mass transport, as seen in sensitivity analyses, it was theorized that by assessing areas with moderate rainfall and assessing meteorological stations by wind conditions, representative high-end and central tendency sites could be determined. In general, light winds will yield high concentrations and high dry deposition values. Dry deposition of particles is the product of deposition velocity and concentration. Based on the equation for concentration from area source emissions, wind speed is inversely proportional to concentration values (U.S.EPA, 1995a). Predominant winds will also lead to higher long term average concentrations.

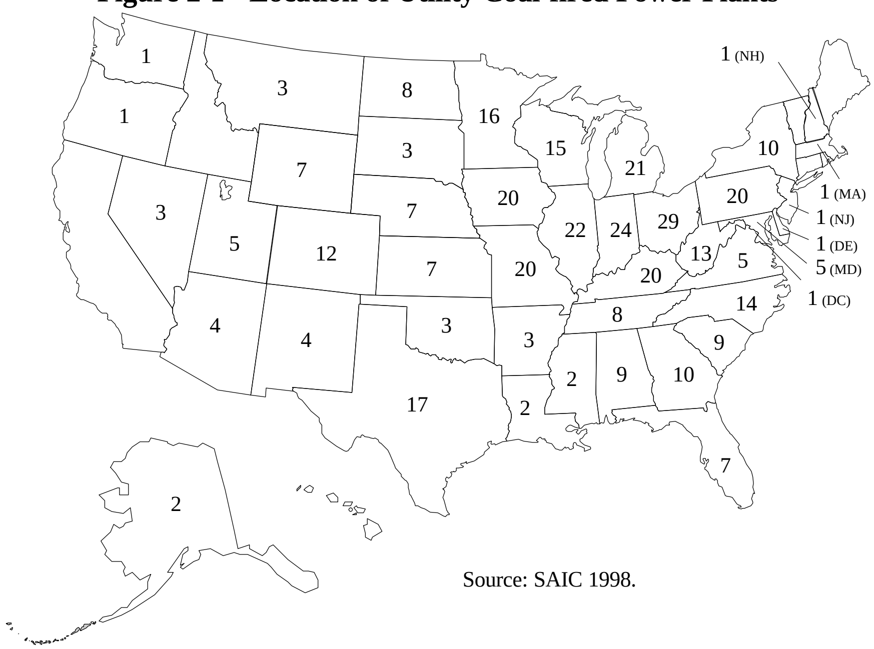
Using this selection process, distribution data for concentration, windroses, and climatological data from 29 meteorological stations were examined. These 29 meteorological stations were used to represent the conditions throughout the continental United States in a previous analysis (EQM, 1993) and were selected to represent a range of meteorological and physiographic conditions. The high-end meteorological location was selected based on its high particulate concentration values, light predominant winds, and moderate precipitation. The central-tendency meteorological location was selected based on its moderate concentration values, moderate predominant winds, and moderate precipitation. The meteorologic data that were used in this analysis are presented in Table 2-1.

These data sources and conditions supplied the guidelines for selection of input parameters that were used in the deterministic risk analysis that was performed. A description of this deterministic analysis is presented below.

Table 2-1. Meteorological Data Used in Analysis

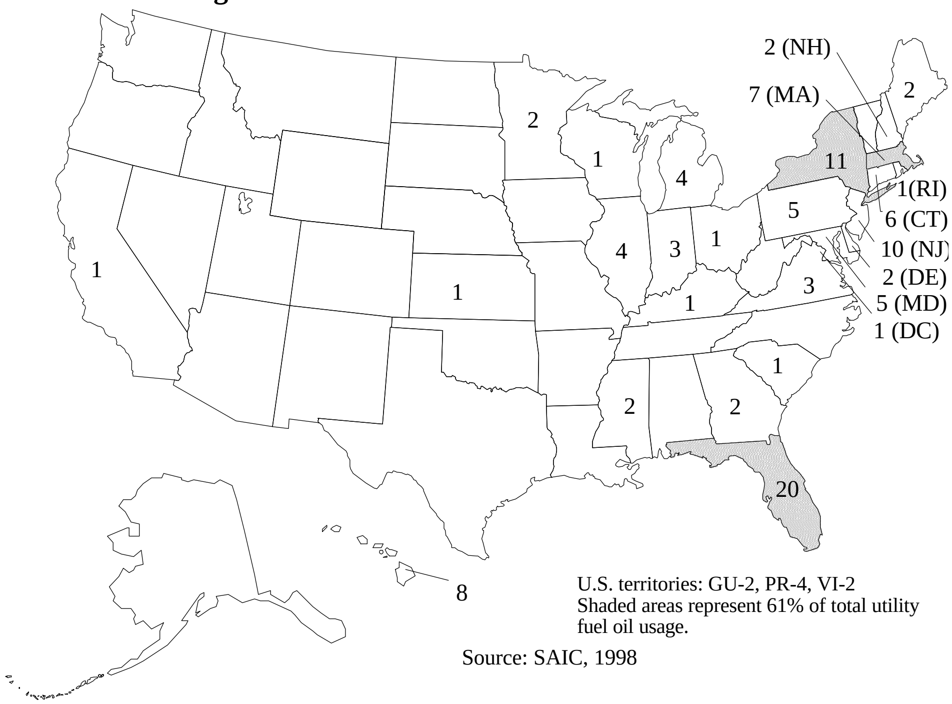
Parameter	Units	High-end	Central Tendency
Average Annual Precipitation	inches/yr	41.4	15.4
Average Annual Precipitation	cm/yr	105.2	39
Average Annual Evapotranspiration	cm/yr	48	25
Average Annual Runoff	in/yr	8.8	0.5
Average Annual Runoff	cm/yr	22	1
Average Annual Temperature	F	55	44.4
Average Annual Temperature	K	286	280
Mean Annual Wind Speed	m/s	4.6	6.2
USLE Rainfall/ Erosivity Factor	1/yr	189	58

Figure 2-1 - Location of Utility Coal-fired Power Plants



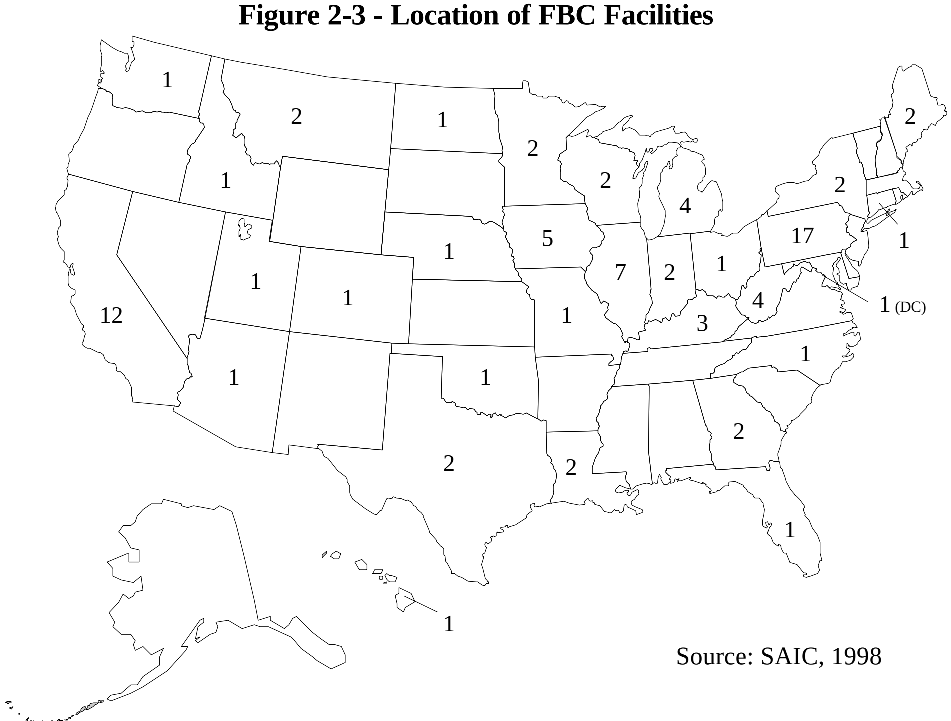
Source: SAIC 1998.

Figure 2-2 - Location of Oil-fired Power Plants



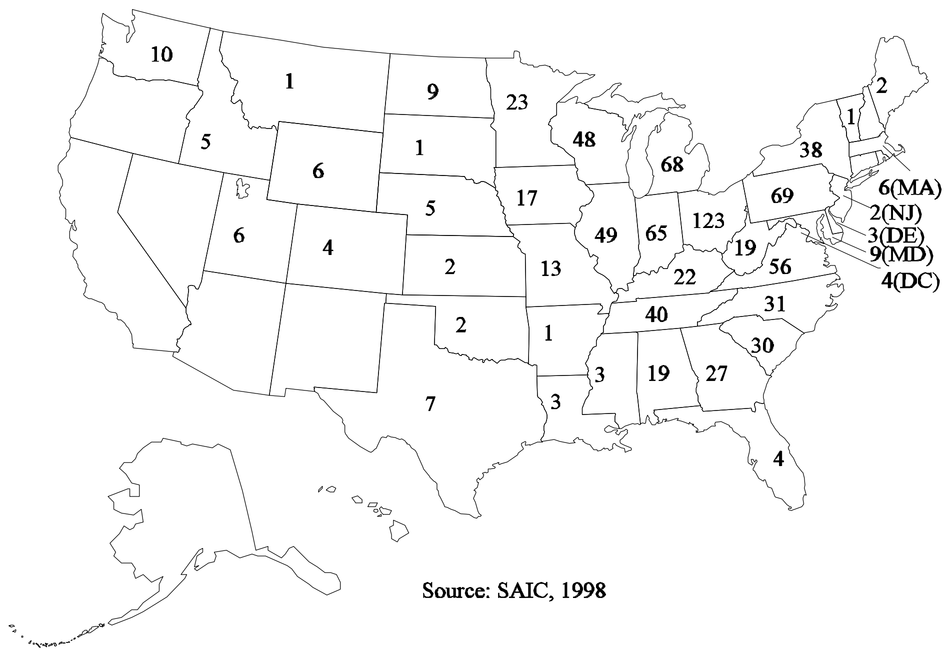
Source: SAIC, 1998

Figure 2-3 - Location of FBC Facilities



Source: SAIC, 1998

Figure 2-4 - Location of Non-utility Facilities



2.2 Risk Assessment Methodology

The risk assessment was conducted in accordance with the EPA human health and ecological risk assessment guidance and is consistent with the methodology employed in recent OSW listing projects including the *Risk Assessment Support for Cement Kiln Dust Regulatory Activities* (RTI, 1997) (henceforth referred to as the CKD Analysis) and the *Nongroundwater Pathway Risk Assessment; Petroleum Process Waste Listing Determination* (RTI, 1997). These risks were developed using a deterministic method, which produces point estimates of risk based upon single values for input parameters.

2.2.1 Bounding Analysis

The risk assessment began with a bounding analysis. Bounding analyses estimate risk for the most exposed individual and exposures for the most exposed ecological receptors. For a deterministic analysis, all parameters are set at high-end values (5^{th} or 95^{th} percentile depending on the parameter's correlation with risk) and human health risks or hazard quotients are calculated. From these same model runs, maximum media concentrations are obtained and used in the ecological risk assessment. Constituents, waste streams, or waste management units/disposal practices that show no excess risk (defined as carcinogenic risk values greater than or equal to 1E-6, human health hazard quotients greater than or equal to 1.0 for non-carcinogens, or ecological hazard quotients greater than or equal to 1.0) in the bounding analysis can be excluded from further analysis. For constituents that do not bound out, input parameters are refined.

2.2.2 High-end/Central Tendency Analysis

High-end estimates of individual risk are intended to represent risk above the 90th percentile of the distribution of individual risk in a population but not higher than the individual with the highest risk. Exposure estimates for the ecological high-end scenario are presumed to represent above the 90th percentile exposure in the ecosystem. Central tendency estimates are intended to represent risks near the 50th percentile of the distribution of individual risk or, in other words, the risk to someone with a more typical exposure. Similarly, central tendency exposure estimates for ecological receptors are presumed to represent more typical exposures in the ecosystem.

For the central tendency risk estimation, all parameters are set to their central tendency value and the risk or hazard quotient for each constituent is calculated. The resultant media concentrations are then used in the ecological risk assessment. The high-end risk estimation utilizes a double-high-end assessment methodology. With this approach, two parameters at a time are set to their high-end value while the remainder of the parameters are set at central tendency and risk values are calculated. Then another combination of parameters are set at their high-end value while the remainder are set at central tendency and this set of results is calculated. This continues until risk values are calculated for all possible high-end parameter combinations. For completeness, and to gain insight on the importance of each parameter individually, each parameter is independently set to high-end while all other parameters are set at central tendency and risks are calculated. The maximum media concentrations from these model runs are then used in the ecological risk assessment.

In depth discussions of the human health and ecological risk assessments are presented in Sections 5 and 6 respectively.

3.0 WASTE STREAM AND WASTE MANAGEMENT UNIT CHARACTERIZATION

3.1 Waste Streams and Waste Management Units Modeled

Waste stream and waste management unit characteristics are the basis for all potential contaminant releases. All waste characterization data and waste management data were supplied by EPA.

The following sections discuss the types of waste and waste management units that were determined to exist for each of the industry sectors assessed in this analysis.

It should be noted that many of these waste streams are managed in surface impoundments. The only non-groundwater pathway for active surface impoundments is volatilization. Because the constituents of concern consist only of non-volatile metals, it was determined that an active surface impoundment would not result in emissions and human health exposures for non-groundwater pathways.

3.1.1 Utility Coal Co-managed Wastes

As discussed above, the analysis for the 1993 Regulatory Determination focused on the four large volume wastes that are produced from coal combustion - fly ash, bottom ash, boiler slag, and FGD sludge. The analysis also only considered the electric utility industry. This analysis assesses the human health and ecological risks from the co-management of these large volume wastes with smaller volume wastes such as pyrites, demineralizer regenerate, coal mill rejects, and boiler chemical cleaning wastes.

It was determined from the Coal Combustion By-products and Low Volume Wastes Co-management Survey data base (EPRI, 1997) that the majority of these wastes are managed in onsite solid waste landfills, surface impoundments, or are used in ways defined as "beneficial uses." Examples of beneficial uses for utility coal co-managed wastes include: minefill and mine reclamation; aggregate in concrete; asphalt and grout; waste stabilization; road materials; roofing shingles; and as an ingredient in many other products.

3.1.2 Utility Oil Combustion Wastes

The total volume of utility oil combustion waste is small compared with the quantity generated by the coal-fired utilities. This difference is due to a smaller number of oil-fired utilities and a lower ash content of fuel oil as opposed to coal. Typically, 70 percent of oil combustion waste is fly ash and 30 percent is bottom ash.

Utility oil combustion wastes are managed in landfills and surface impoundments (commonly called solids settling basins in the industry). Surface impoundments, however, are rarely the final disposal unit. Instead, these wastes typically remain in surface impoundments only temporarily. Then, the solids are dredged and transported to an off-site landfill or utilized for vanadium recovery. Thus, a given oil combustion waste stream may be managed in both a surface impoundment and a landfill in the course of waste management.

3.1.3 FBC Wastes

Two types of waste are generated in FBC units - fly ash and bed ash. Thus, there are three potential wastes to be managed at FBC facilities - fly ash, bed ash, and combined ash. This analysis models combined ash wastes because it more accurately represents the waste as it is managed.

FBC wastes are either disposed of or employed in beneficial uses. Disposal is limited almost exclusively to onsite, permitted solid waste landfills. Beneficial uses of FBC waste include mine reclamation, agricultural applications (such as liming agent), cement uses (raw feed or cement substitute), other geotechnical uses, and stabilization applications.

3.1.4 Non-utility FFC Wastes

The non-utility FFC universe includes both coal- and oil-fired boilers. Steam generated by non-utility combustors is used to generate electricity, to provide heat, or as a production process input. Because they use similar combustion technologies, the types of combustion wastes (e.g., fly ash, bottom ash) generated by non-utilities are the same as those generated by utilities. Based on a survey by the Council of Industrial Boiler Owners (CIBO), it also appears that co-management of high-volume wastes with low volume wastes is as common with non-utility coal combustion as with the utility coal combustion. However, EPA did not receive waste characterization data for non-utility FFC combustion facilities. Therefore, the waste characterization data used to assess risks from utility coal-fired co-managed wastes are used to characterize waste for the non-utility coal combustion sector.

Disposal in landfills is the primary disposal practice for non-utility coal combustion residuals. Non-utility combustion waste are disposed of in both onsite landfill and offsite commercial landfills.

3.2 Waste Management Unit Dimensions

Waste management unit sizes were derived by Sciences Applications International Corporation (SAIC). This section summarizes the derivation of these dimensions and is excerpted from SAIC 1998b. Where practicable, a reality check was conducted by comparing parameters that are calculated based on the hypothetical WMU to those parameter values from other data sources. This assured that internal consistency was maintained and that the hypothetical WMU's were realistic. FFC wastes are managed in a variety of settings and therefore this report uses different assumptions for each management practice (e.g., landfill) in each of the generating sectors (e.g., oil combustion, FBC). Specifically, two different active lives are used in this report, 30 years and 40 years, which refer to the time that the management unit is open and accepting waste. These two active lives are based on two separate data sources. The 40 year active life is

an assumption based on discussions with EPRI concerning typical active lives of management units at utilities. Therefore, all management units similar to these types of units were assigned values of 40 years: onsite landfills and impoundments at coal-fired comanagement utilities and at FBC utilities. Commercial offsite landfills, however, are not similar to these types of units and therefore different data were used. Specifically, EPA
estimates the lifetime of an offsite commercial landfill to be approximately 30 years (Additional Listing Support Analyses for the Petroleum Refining Listing Determination, EPA, 1998). WMU's for which specific data on lifetime were not available were assumed to have a lifetime of 30 years.

3.2.1 Utility Coal-fired Co-managed Waste Dewatered Surface Impoundment

For the non-groundwater pathways the only WMU parameter that is of concern is area. The following areas were used for contaminant fate and transport modeling:

- a surface area of 364,000 square meters, or 90 acres, corresponding to the 50th percentile value reported in the EPRI comanagement survey;
- a surface area of 1,670,000 square meters, or 412 acres, corresponding to the 95th percentile value reported in the EPRI co-management survey.

3.2.2 Utility Coal-fired Co-managed Waste Onsite Landfill

3.2.2.1 <u>Central Tendency</u>. The following data were used for this scenario:

- a surface area of 267,000 square meters, or 66 acres, corresponding to the 50th percentile value reported in the EPRI co-management survey; and
- a height of 9.45 meters, or 31 feet, corresponding to the 50th percentile value derived from capacity and area reported in the EPRI co-management survey.

These modeled dimensions correspond to the following assumptions about the hypothetical landfill:

- a total capacity of 3,300,880 cubic yards; and
- an annual waste disposal rate of 82,522 cubic yards, assuming a 40 year lifetime.

For comparison purposes, the annual waste disposal rate and total capacity are just smaller than the median values (40th and 47th percentiles, respectively) calculated for the landfills in the EPRI co-management survey. Thus, the hypothetical landfill is well within the range of dimensions reported in the EPRI population.

3.2.2.2 <u>High-end</u>. The following data were used for this scenario:

- a surface area of 1,330,000 square meters, or 328 acres, corresponding to the 95th percentile value reported in the EPRI comanagement survey; and
- a height of 9.45 meters, or 31 feet, corresponding to the 50th percentile value derived from capacity and area reported in the EPRI comanagement survey.

These modeled dimensions correspond to the following assumptions about the hypothetical landfill:

- a total capacity of 16,404,373 cubic yards; and
- an annual waste disposal rate of 410,109 cubic yards, assuming a 40 year lifetime.

For comparison purposes, the annual waste disposal rate and total capacity are near the upper end but within the distribution of values reported in the EPRI survey (80th and 86th percentile, respectively).. Thus, the hypothetical landfill is well within the range of dimensions reported in the EPRI population.

3.2.3 Utility Oil-fired Waste Onsite Landfill

Because no directly reported data on the dimensions of oil combustion waste landfills were available, the dimensions of the hypothetical landfills were derived differently from those for other FFC waste management units. One facility in the EPRI oil ash report disposes of 3,000 tons per year of oil ash in an on-site landfill. This annual waste disposal rate is at the upper bound of the distribution of generation rates in the EPRI report. It was decided that there were not sufficient data to characterize a high-end and central tendency landfill and thus only one size was modeled. A density of 1 ton/cubic yard and constant disposal rates over a 30 year life for the unit to arrive at the following statistics for a hypothetical landfill:

- an annual waste disposal rate of 3,000 cubic yards;
- a total capacity of 90,000 cubic yards, assuming a 30 year lifetime;
- a surface area of 10,500 square meters, or 2.6 acres; and
- a height of 6.5 meters, or 21 feet.

3.2.4 FBC Onsite Landfill

3.2.4.1 <u>Central Tendency</u>. The following data were used for this scenario:

- a surface area of 155,000 square meters, or 38 acres, corresponding to the 50th percentile value reported in a combined data set from the CIBO survey and the EPRI co-management survey; and
- a height of 15.8 meters, or 52 feet, corresponding to the 50th percentile value derived from capacity and area reported in the combined data set.

These modeled dimensions correspond to the following assumptions about the hypothetical landfill:

- a total capacity of 2,792,747 cubic yards; and
- an annual waste disposal rate of 69,819 cubic yards, assuming a 40 year lifetime.

For comparison purposes, the annual waste disposal rate and total capacity are near the upper end but within the distribution of values reported in the combined data set (66th and 78th percentile, respectively). Thus, the hypothetical landfill is within the range of dimensions reported in the population.

3.2.4.2 <u>High-end</u>. The following data were used for this scenario:

- a surface area of 317,000 square meters, or 77 acres, corresponding to the 95th percentile value reported in a combined data set from the CIBO survey and the EPRI comanagement survey; and
- a height of 15.8 meters, or 52 feet, corresponding to the 50th percentile value derived from capacity and area reported in the combined data set.

These modeled dimensions correspond to the following assumptions about the hypothetical landfill:

- a total capacity of 6,459,787 cubic yards; and
- an annual waste disposal rate of 161,485 cubic yards, assuming a 40 year lifetime.

The annual waste disposal rate is larger than the median, but within the distribution of values reported in the CIBO survey (80th percentile). However, the total capacity is slightly larger than the largest landfill reported in the combined data set (6,100,000 cubic yards). Thus, while this hypothetical landfill falls slightly outside the range of values observed in the population for this one dimension, it is well within the range for the other dimensions. While unusually large in capacity, it probably is sufficiently realistic for purposes of a conservative bounding scenario.

3.2.5 FBC Agricultural Soil Amendment Field

The agricultural field size used in this analysis is consistent with the agricultural field size used in the CKD Analysis. Because the FBC waste material is directly applied to the agricultural field at prescribed rates and frequencies, the size of the field has no impact on the constituent waste concentration in the soil and subsequent impacted media. It was found that the variation in the size of the field had little impact on stream concentrations and fish ingestion risks. Therefore, size of the agricultural field was not varied. The area of the source was determined from the census of agriculture. The field size is the average agricultural field size in states that have acid soils and active cement kiln plants. It is noted that this selection protocol may preclude some states that may apply FBC wastes from being represented, however, it is believed that this would not significantly bias the results of the analysis. The field size that was used in this analysis was 902,450 m^2.

3.2.6 Non-utility Coal-fired Waste Onsite Landfill

3.2.6.1 Central Tendency. Because no directly reported data on the dimensions of non-utility FFC waste landfills were available, the dimensions of the hypothetical landfills were derived differently from those for other FFC waste management units. The central tendency scenario used the 50th percentile annual coal consumption rate from the US90 database to derive an annual ash generation rate, assuming a density of 1 ton/cubic yard. A constant disposal rates over a 30 year life for the unit was used to arrive at the following statistics for a hypothetical central tendency landfill:

- an annual waste generation rate of 1,795 cubic yards;
- a total capacity of 53,844 cubic yards, assuming a 30 year lifetime;
- a surface area of 7,700 square meters, or 1.9 acres; and
- a height of 5.3 meters, or 17.4 feet.

3.2.6.2 High-end. The high-end scenario used the same procedure described above with the 95th percentile annual coal consumption rate to arrive at the following dimensions:

- an annual waste disposal rate of 20,026 cubic yards; and
- a total capacity of 600,782 cubic yards, assuming a 30 year lifetime.
- a surface area of 34,500 square meters, or 8.5 acres; and
- a height of 5.3 meters, or 17.4 feet.

3.2.7 Non-utility Coal-fired Waste Offsite Commercial Landfill

The offsite commercial landfill dimensions were calculated from a subset of the Industrial D database. The Industrial D data were collected in the 1980s to characterize on-site landfills. To make these data more appropriate for FFC waste management, only the landfills from industries most likely to generate FFC wastes are included. The seven industries considered correspond to SIC codes 20, 22, 26, 28, 33, 37, and 49 (an eighth industry, SIC code 82, was initially considered but no Industrial D landfill data existed for this sector). These seven industries use the largest quantities of coal for non-utility fuel and were therefore assumed to represent on-site FFC waste disposal practices. The 50th percentile median area associated with these facilities is 34,400 square meters, while the 95th percentile high-end area is 404,700 square meters.

The landfill capacity was calculated based on a regression equation presented in EPA's *Air Characteristics Study* (U.S. EPA, 1998). The landfill areas were estimated to be 96,600 yd^3 and 1,100,000 yd^3 for the central tendency and high-end landfills respectively. Using the annual waste generation rate that was calculated for the onsite landfill and an assumed 30 year landfill lifetime, a waste fraction (percentage of landfill that is FFC waste) was calculated. Waste fractions were 0.56 for the central tendency landfill and 0.55 for the high-end landfill. The starting waste concentration is then multiplied by the waste fraction to derive an effective starting concentration for the fate and transport modeling.

4.0 FATE AND TRANSPORT IN THE ENVIRONMENT

This section describes the types of releases that can occur from each WMU assessed in the non-groundwater analysis and the models that were used to estimate constituent fate and transport in the environment. Detailed discussions of the models used for estimating fate and transport are provided in the Appendices. Appendix C details the overland transport models used to calculate soil erosion and run-off. Appendix D gives an overview of the air dispersion modeling that was performed. Appendices E and F provide all equations used for the Indirect Exposure Methodology (IEM) modeling. Appendix E equations are used to model fate and transport, exposure, and risk from landfills and the dewatered surface impoundment while Appendix F provides the equations used to model fate and transport, exposure, and risk from the agricultural soil amendment scenario. Appendix G provides the model used to calculate contaminant losses in the agricultural field to arrive at the starting concentration for the soil amendment scenario.

4.1 Waste Management Unit Releases

This section explains the types of environmental releases that can be expected from each of the waste management options described above. Also, for each of the waste management options, the major assumptions that were used for the fate and transport modeling are presented.

Landfill

Potential contaminant releases from landfills include: leaching to groundwater; overland transport from erosion and runoff; and air emissions. A complete description of the groundwater pathway can be found in the SAIC Background Document. The overland transport mechanisms assessed in this analysis are depicted in Figure 4-1. Both erosion and runoff transport contaminants from the source and deposit them onto the intervening area, the agricultural field, and the stream as depicted in Figure 4-1.

Air emissions can originate from waste unloading operations, spreading and compacting operations, resuspension of particulates from vehicular traffic, and from wind erosion.

All onsite landfills are assumed to contain only the FFC waste being assessed. The landfill sizes were provided by EPA and are discussed in Section 3.2.

Surface Impoundment

The primary release mechanisms for active surface impoundments are leaching to groundwater and air emissions of volatile constituents. Since no volatile emissions are assumed and a separate groundwater analysis is being performed, an active surface impoundment was not

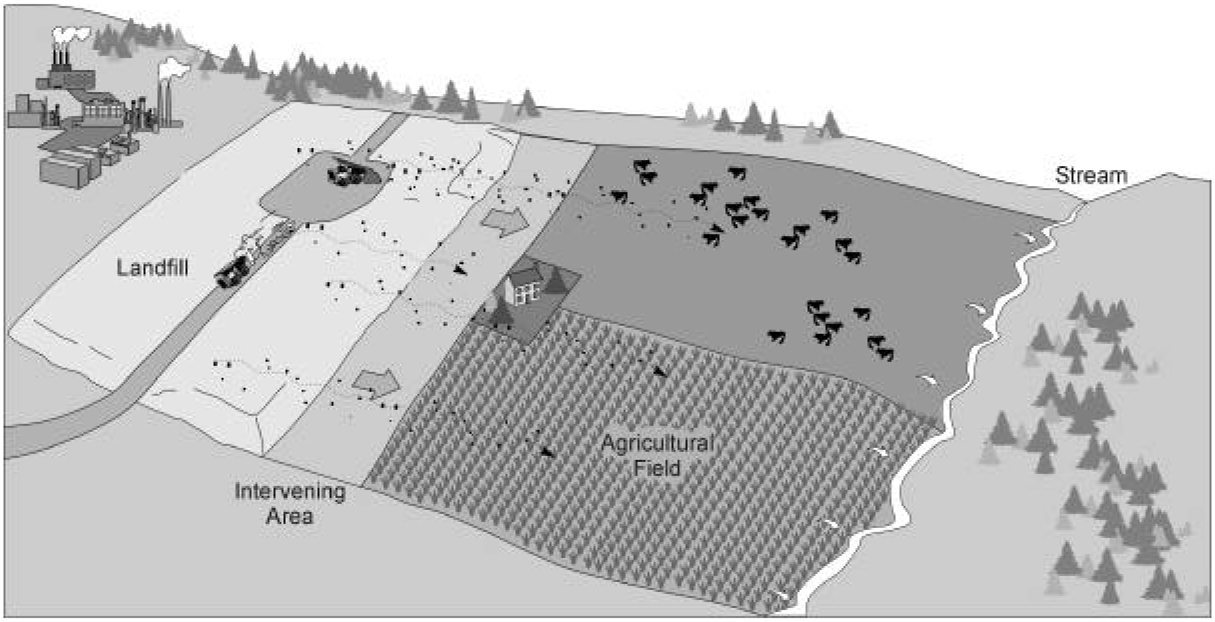


Figure 4-1. Depiction of Environmental Release and Transport for the Landfill Scenario

modeled for human health risks.[2, 3] Rather, a closed, dewatered surface impoundment was modeled for aboveground risk. A dewatered surface impoundment is assumed to be below grade due to design characteristics. Therefore, it is assumed that there would be no overland transport of contaminants through erosion and runoff. And since there are no waste loading operations, it follows that the only source of emissions will be from wind erosion. Emissions were estimated and dispersion was modeled as a flat, even-grade unit. These assumptions will tend to overestimate risks from this type of unit. A dewatered surface impoundment is depicted in Figure 4-2.

Beneficial Uses

There are a large number of beneficial uses identified for these wastes. Most of these beneficial uses employ the wastes as ingredients in a final product (e.g., concrete, asphalt, roofing shingles) and thus the wastes are considered encapsulated and the mobility of its constituents in the environment is greatly reduced. EPA also limited its review to those beneficial uses where the waste is applied directly to the ground. These uses include minefill/mine reclamation and use of the waste as agricultural soil amendment. Application of the waste to agricultural lands as a soil

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[2] No volatile emissions are assumed due to the fact that only non-mercury metals are assessed.

[3] An active surface impoundment was however modeled for ecological risks. A discussion of this analysis is presented in Section 6.0, and the characterization of the surface impoundment is discussed in Appendix B.

amendment may provide risks from a variety of pathways.[4] Use of the waste as minefill/mine reclamation also may have significant human health and ecological impacts. However, the data provided by EPA indicate that minefills are less numerous and generally smaller than conventional landfills. Since the mine would be modeled like the landfill, a separate analysis was not performed to assess risks from FFC wastes used as minefill.

Once applied to the agricultural lands, contaminants are taken up by the plants grown on the amended soil. Wind erosion and runoff/erosion from the agricultural field to a nearby stream will also occur. This source is depicted in Figure 4-3.

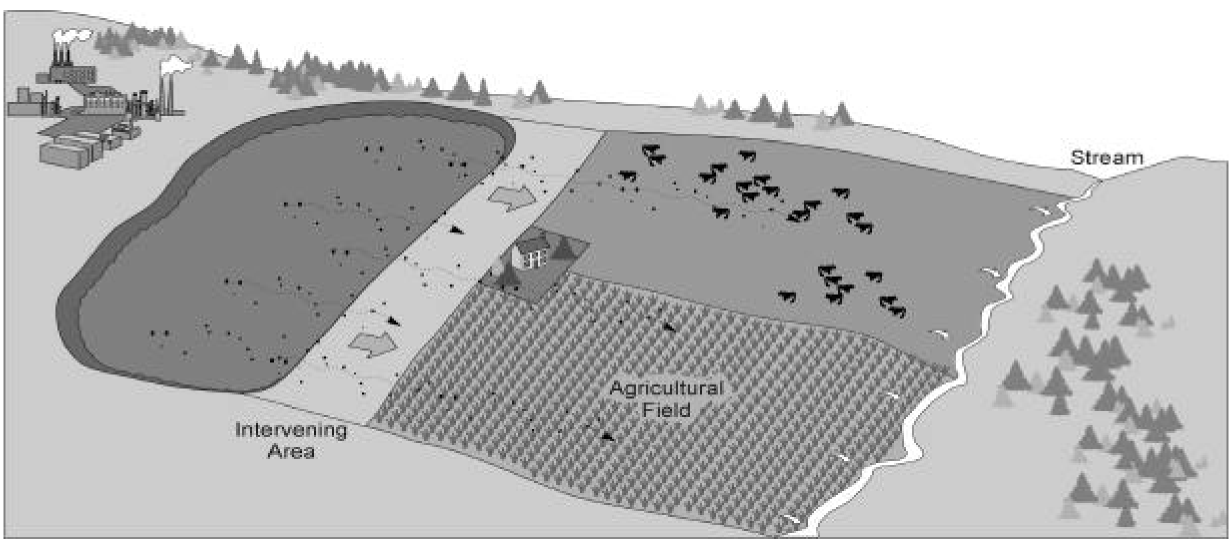

Figure 4.2 Depiction of Environmental Release and Transport for the Dewatered Surface Impoundment Scenario

[4] FBC wastes are used as liming agents due to their naturally high alkilinity.

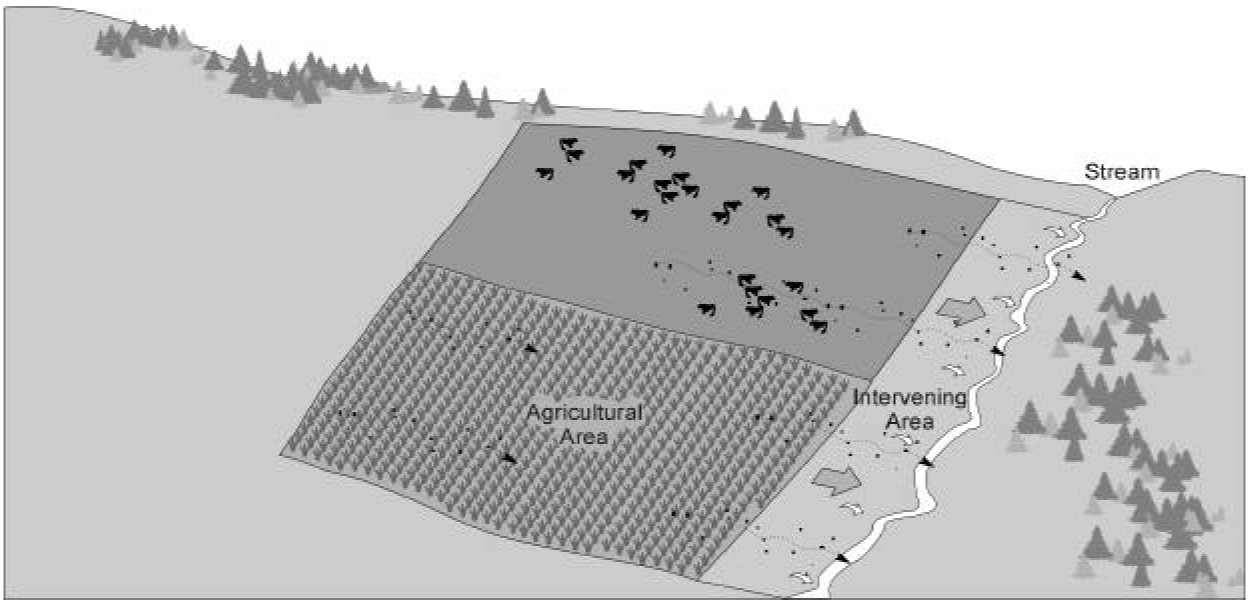

Figure 4-3. Depiction of Environmental Release and Transport from the Agricultural Soil Amendment Scenario

4.2 Soil Erosion and Runoff

Soil erosion and runoff were assumed to occur from all landfill scenarios and from the soil amendment scenario. A modified Universal Soil Loss Equation (USLE) was utilized to estimate overland erosion and runoff. The USLE is an empirical erosion model originally designed to estimate long-term average soil erosion losses to a nearby waterbody from an agricultural field. The USLE was modified to estimate soil erosion and runoff from WMU's, across intervening areas, to nearby waterbodies, by evaluating this process in an integrated setting. A complete write-up of the overland transport models is provided as Appendix C.

4.3 Particulate Air Emissions and Dispersion

Particulate emissions differ between each of the WMU/sources that were assessed in this analysis. Therefore, emissions from each of the WMU/sources will be discussed separately below.

Dispersion for all WMU/sources was modeled using the EPA's Industrial Source Complex Short Term (ISCST3) Model. ISCST3 can be used for continuous releases from one or more point, area, volume, or line sources. For all sources, wet particle deposition, dry particle deposition, and particulate air concentration were modeled for receptors located 300 meters from the source for central tendency and 75 meters from the source for high-end. Thirty-two, equal spaced, receptor points were modeled for each distance. (See Figure 4-4 for depiction of distance to receptor and spacing of modeled receptors.) Additional detail on ISCST3 is presented in Appendix D.

4.3.1 Particulate Air Emissions and Dispersion from Landfills

Several different types of particulate air emissions contribute to aggregate particulate emissions from landfills. These include emissions from wind erosion of the active landfill, from vehicle resuspension of contaminants while delivering waste to the landfill, from waste material blowing from trucks hauling the waste to the landfill (assume less than 100% cover control efficiency), from unloading the waste at the landfill, and from spreading and compacting operations. Figure 4-4 presents a pictorial representation of the landfill.

Emissions from wind erosion are estimated only for the active cell of the landfill. It is assumed that only part of the entire landfill will be active at any particular time and that the rest of the landfill will have a cover of uncontaminated, native soil. The area of the active cell is assumed to be one years worth of waste capacity. Therefore, the active landfill area is equal to the total landfill area divided by the active life of the landfill. Wind erosion emissions are calculated using the same methodology employed in *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (U.S. EPA, 1990). For dispersion modeling, the active cell is assumed to be located in the center of the landfill (see Figure 4-4) Using the center cell will yield results that simulate an average concentration over the entire lifetime of the landfill for any given receptor.

Both emissions from vehicle resuspension and from waste being blown from the trucks are estimated using AP-42 emissions equations (U.S. EPA, 1995b). AP-42 equations are empirically derived and are used to estimate fugitive dust emissions from a variety of activities for both point and area sources. For dispersion modeling, the source of these emissions was assumed to be the area of the road depicted in Figure 4-4. The length of the road is dependent on the size of the landfill being modeled while the width of the road is set at three meters for all landfills. Three meters was determined to be wide enough to allow one way traffic under the assumption of a single entrance and exit for the landfill. A sensitivity analysis showed that maximum air concentration results are slightly sensitive to the orientation of the landfill. Higher air concentrations were calculated when the landfill was oriented with the predominant wind directions as depicted in Figure 4-4.

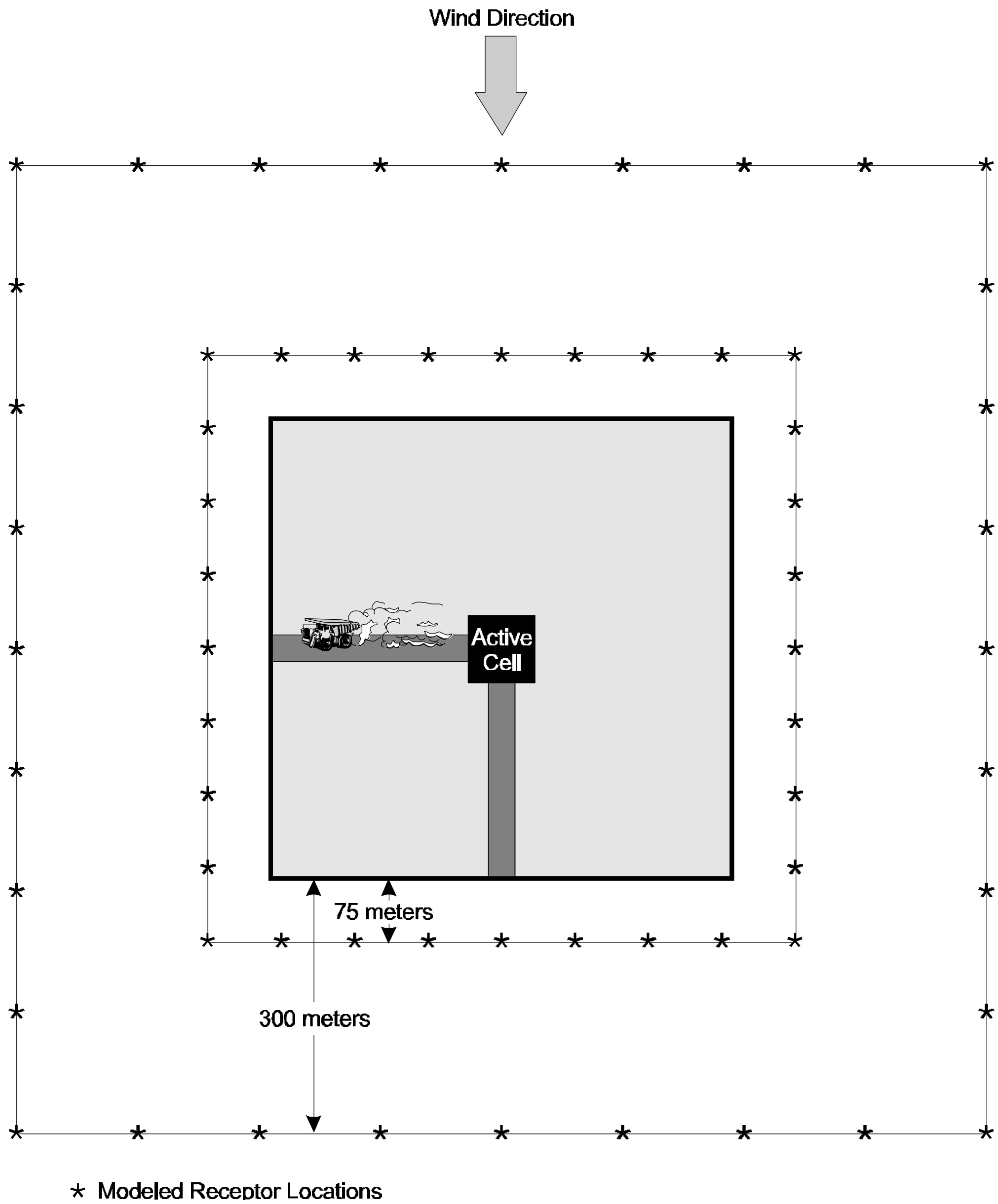

Figure 4-4. Depiction of Landfill Scenario Air Emissions and Dispersion

4.3.2 Particulate Air Emissions and Dispersion from the Dewatered Surface Impoundment

Only emissions from wind erosion were estimated for the dewatered surface impoundment. These emissions were estimated using the same methodology as was employed in the landfill scenarios. Dispersion was modeled for the entire surface area of the waste unit. Receptors were modeled as they were for the landfill scenarios.

4.4 Fate and Transport from the Agricultural Field

Because FBC residues contain alkaline oxides and plant nutrients that are useful in agriculture, they are often applied to agricultural lands to increase pH and supply these necessary nutrients (USDA, 1988). Cement kiln dust (CKD) is also used for this purpose and this practice is the focus of a recent risk analysis to assess the risks that result from applying CKD to agricultural lands as a liming agent. Due to these inherent similarities, the development of this scenario and the subsequent modeling was based on the methodology employed in the CKD Analysis. For this scenario it is assumed that the FBC combined ash is applied directly to an agricultural field as a liming agent. Therefore, the same input parameters were used for this analysis that were used in the CKD Analysis. These values are presented in Table 4-1. It is also assumed that FBC wastes will be used, as CKD wastes are used, on acidic soils. Therefore, the same soil parameters and meteorological locations that were used in the CKD Analysis are used to estimate risks from FBC wastes used as agricultural soil amendment.

The same routes of exposure that were assumed in the CKD Analysis and the other scenarios for this analysis are assumed for this scenario. These exposure routes include soil ingestion, ingestion of plants, beef, and dairy products grown on the soil amended agricultural field, and ingestion of fish caught in a stream nearby the soil amended agricultural field. Inhalation risks to the farmer from tilling and wind erosion were assessed in the CKD Analysis. These risks bounded out and were considered inconsequential relative to risks from other pathways. A comparison of constituents and starting concentrations between CKD and FBC revealed that the FBC risks from inhalation would likely be insignificant. Therefore, it was assumed that risks would be driven by other pathways and thus inhalation risks were not evaluated for this analysis.

Partitioning of metals in the agricultural field was estimated using a set of equations that calculate contaminant losses due to leaching and rainwater runoff. The model tracks the average annual soil concentration for a period of 100 years of active use (corresponding to the period over which waste application occurs) followed by 40 years of inactive use. The maximum rolling average starting soil concentration which is based on the exposure duration is then used as the soil concentration for all pathways. A complete write-up of the partitioning equations used for this analysis is presented in Appendix G.

Crops on the agricultural field receive dry deposition of particles from the agricultural field as well as uptake of contaminant through the roots.

The stream also receives runoff/erosion, and air deposition from the agricultural field. The equations used for overland transport from the agricultural field to the stream are provided in Appendix F. The air deposition to the stream is set equal to the onsite deposition of dry particles to the agricultural field. It is assumed that the stream is 75 meters from the agricultural field.

Table 4-1. Input Values for Agricultural Liming Practice Parameters

Parameter	**Central Tendency**	**High-end**
Application Rate (tons/acre/application)	3	5
Application Frequency (years)	1/3	1/2
Tilling Depth (cm)	15	10

5.0 HUMAN HEALTH RISK ANALYSIS

5.1 Constituents of Concern

The constituents that were modeled were based on analytical data provided by EPA. The constituents with human health toxicity values (i.e., RfC, RfD, or CSF values) that were assessed for the human health risk analysis are provided in Table 5-1. A complete description of the waste characterization for each waste stream is provided in Appendix B.

Table 5-1. Constituents Included in the Human Health Risk Analysis by Wastestream

Wastestream	Constituents Modeled for Human Health
Coal-fired utility and non-utility co-managed wastes	Nickel, silver, thallium, arsenic, barium, beryllium, boron, cadmium, chromium VI, cobalt, selenium
Oil-fired utility wastes	Nickel, silver, thallium, arsenic, barium, boron, cadmium, chromium VI, cobalt, copper, vanadium, zinc
FBC wastes	Nickel, silver, thallium, arsenic, barium, beryllium, boron, cadmium, chromium VI, cobalt, selenium

5.2 Potential Routes of Exposure

Individuals can be exposed to the previously described contaminant releases through a number of exposure routes. The relative contribution of any one of these exposure pathways for a particular chemical is dependent on the physical and chemical characteristics of the chemical, the properties of the wastestream, and the environmental setting that is being modeled. The following exposure pathways were assessed for this analysis.

Groundwater Ingestion

For any of the waste management units described above, contaminants can leach from the waste into the groundwater and subsequently be ingested by individuals who rely on well water. A detailed discussion of the modeling of and risks from this pathway can be found in the SAIC Background Document. Untreated surface water is not considered a source of residential drinking water. Further, it is assumed that exposure from inadvertent ingestion of surface water would be in small volume and infrequent.

Air Emissions, Inhalation

Another source of exposure is from inhalation. Since this analysis only considers non-mercury metals, only emissions of particulates are considered. For the inhalation pathway only emissions and air concentrations of respirable PM10 are considered.

Ingestion of Contaminated Soil

Exposure from this pathway stems from inadvertent ingestion of contaminated soil. The consumed soil has been contaminated by air deposition and deposition from overland transport of contaminants through erosion and runoff. For the soil amendment scenario, exposure stems from inadvertent ingestion of soil directly contaminated from application of the waste product.

Ingestion of Fruits and Vegetables

Individuals may ingest fruits and vegetables that have been contaminated by migration of waste constituents to offsite agricultural fields or from direct application of the waste as agricultural soil amendment. For above-ground fruits and vegetables, this pathway includes two concurrent mechanisms of plant uptake: air deposition directly to the plant, and deposition to soil (including air particulate deposition and erosion and runoff deposition) followed by uptake by the plant. For root vegetables, this pathway includes only one mechanism of plant uptake: deposition to soil followed by root uptake by the plant. Uptake via direct deposition to the root plant is not considered, since the edible (root) portion is protected from contact with contaminants in air, and it is unclear whether contaminants deposited on plant surfaces are internalized by plants.

Beef and Dairy Products Ingestion

Individuals may also ingest beef and dairy products from cows raised on contaminated agricultural fields. The estimated constituent concentrations in beef and dairy products are estimated based on the dietary intake assumptions for cattle. The cattle's diet is assumed to consist of forage (i.e., pasture grass and hay), silage, and grain. In addition, the cattle are assumed to directly ingest the contaminated soil.

Fish Ingestion

Exposure for this pathway stems from human ingestion of fish that are caught in a stream located nearby a waste management unit or soil amended agricultural field. The stream has been contaminated from runoff and erosion from the waste management unit and direct deposition of particles into the stream. Once the constituents are in the surface water, fish may "ingest" the dissolved constituents during normal respiration across the gills. In addition, contaminants that concentrate in the sediments may be introduced into fish through the food chain, beginning with benthic organisms (i.e., bottom dwellers). Contaminants can accumulate in fish from either or both of these exposure vehicles depending on the chemical's potential for bioconcentration and bioaccumulation.

There is the potential for contaminated groundwater recharge to surface water bodies (i.e., streams) resulting in contamination of surface waters and fish, and risk from subsequent fish ingestion. This pathway was not considered in this analysis for several reasons. The primary reasons are that this scenario is a very site specific phenomena and would require site specific data to model. This type of analysis is not consistent with the more general bounding analysis performed here. Also, it is assumed that the load to the surface waterbody from groundwater recharge would be far surpassed by the load from the overland routes (i.e., erosion and runoff) or from outfall permits under the National Pollutant Discharge Elimination System (NPDES).

5.3 Identification of Receptors

Receptors evaluated in multiple pathway analyses are selected to represent a typical individual and highly exposed individuals. Highly exposed subpopulations (i.e., high-end receptors) are examined to ensure that significant subpopulations of potentially highly exposed individuals are protected. For human receptors, potentially more exposed populations are identified based on the behavior of potential receptors such as dietary habits and activities (e.g., consumption of self-caught fish or growing foods for home consumption) or based on their proximity to a contaminant source. Selected receptors include: (1) adult resident, (2) home gardener, (3) farmer, (4) child of farmer, and (5) fisher. This array of receptors covers all plausible exposure pathways.

An adult resident living near the waste management unit could inhale airborne constituents transported off-site as particulates or ingest constituents via incidental soil ingestion.

Indirect exposure pathways include uptake of constituents from home-grown fruits and vegetables, ingestion of home-grown meat and dairy products, and ingestion of fish caught from local impacted streams. A home gardener, farmer, and fisher were selected to evaluate these indirect exposure pathways.

The home gardener is an adult resident that, in addition to the exposure pathways identified for the adult resident, is also exposed by ingesting contaminated homegrown fruits and vegetables. The farmer and child exposure includes all of the exposure pathways listed above plus ingestion of meat and dairy produced on their farm. The child receptor was included to represent a sensitive subpopulation potentially exposed to all of the most plausible direct and indirect exposure pathways. In modeling the child receptor, the input parameters that differ from the adult farmer receptor include: body weight; ingestion and inhalation rates; and exposure duration.

Finally, an adult fisher was selected to assess potential risk associated with fish ingestion from an impacted stream. Parameters used to estimate impacts to a nearby fishable stream are provided in Table 5-2. The dietary consumption items assessed for each receptor are presented in Table 5-3.

Exposure factors were selected to represent both central tendency (50th percentile) and high-end (95th percentile) values for the selected receptors and exposure pathways. These values were obtained from U.S. EPA's Exposure Factors Handbook (EFH) (U.S. EPA, 1998). Table 5-4 presents ingestion rates and inhalation rates for all receptors. Note that beef and dairy ingestion rates are presented in both whole weight (WW) and dry weight (DW). This is due to the fact that the beef and dairy biotransfer factors for cadmium and selenium are based on dry weight concentrations while all other contaminants' biotransfer factors are based on wet weight concentrations.

Table 5-5 presents the various exposure duration values used. It should be noted that the time in which the child exhibits pica behavior was set at 6 years.[5] After age six, the child ingests soil at the same rate as the adult for a period of years that were in accord with the exposure duration distribution presented in Table 5-5.

The fraction of each dietary item ingested that is contaminated is presented in Table 5-6. Fraction contaminated values are based on data presented in Table 12-71 of the 1998 EFH. This data was used to represent the fraction of the total intake that is assumed to be home grown in the contaminated area. These fractions are mean values and are used as a single quantity in the absence of additional data.

Table 5-2. Waterbody Data Used in Analysis

Parameter	Units	National Average	Reference
Flow	m^3/yr	3.00E+08	HWIR
Total Waterbody Depth	m	0.67	HWIR
Depth of Upper Benthic Layer	m	0.03	US EPA, 1993
Velocity	m/s	0.7	HWIR
Waterbody Area	m^2	1.00E+06	HWIR

[5] Pica behavior is defined as an abnormal desire to eat substances not normally eaten. Often young children will exhibit pica behavior with soil.

Table 5-3. Dietary Consumption Items for Receptor Scenarios

Dietary Item	**Receptor Scenario**				
	Adult Resident	**Home Gardener**	**Farmer**	**Child of Farmer**	**Fisher**
Soil	✓	✓	✓	✓	
Exposed Fruits		✓	✓	✓	
Exposed Vegetables		✓	✓	✓	
Root Vegetables		✓	✓	✓	
Beef			✓	✓	
Dairy Products			✓	✓	
Fish					✓

Table 5-4. Intake Rates Used in Analysis

		Adult Resident		Home Gardener		Farmer		Child of Farmer		Fisher	
	Unit/Percentile	50	95	50	95	50	95	50	95	50	95
Soil	kg/day	0.00005	0.0001	0.00005	0.0001	0.00005	0.0001	0.0002	0.001	NA	NA
Aboveground Vegetables	kg/DW/d`ay	NA	NA	0.004	0.025	0.006	0.03	0.003	0.017	NA	NA
Fruits	kg/DW/day	NA	NA	0.01	0.072	0.019	0.089	0.0085	0.067	NA	NA
Root Vegetables	kg/DW/day	NA	NA	0.005	0.028	0.0067	0.035	0.0042	0.025	NA	NA
Beef	kg/DW/day	NA	NA	NA	NA	0.031	0.13	0.025	0.059	NA	NA
	kg/WW/day	NA	NA	NA	NA	0.11	0.45	0.088	0.21	NA	NA
Dairy	kg/DW/day	NA	NA	NA	NA	0.17	0.63	0.088	0.21	NA	NA
	kg/WW/day	NA	NA	NA	NA	0.73	2.64	0.365	0.88	NA	NA
Air	m3/hr	0.83	NA	0.83	NA	0.83	NA	0.2	NA	NA	NA
Fish	g/day	NA	NA	NA	NA	NA	NA	NA	NA	61.8	393.6

Table 5-5. Exposure Durations Used in Analysis

		Adult Resident		Home Gardener		Farmer		Child of Farmer		Fisher	
Exposure Duration	**Unit/Percentile**	**50**	**95**	**50**	**95**	**50**	**95**	**50**	**95**	**50**	**95**
Exposure Duration (soil ingestion) [a]	year	3.3	32.3	3.3	32.3	10	58.4[b]	6	18	3.3	32.3
Exposure Duration (all other exposures)[a]	year	3.3	32.3	3.3	32.3	10	58.4[b]	7.3	18	3.3	32.3

Table 5-6. Fraction of Dietary Item Contaminated

Dietary Item	Adult Resident	Home Gardener	Farmer	Child of Farmer
Soil [a]	1	1	1	1
Aboveground Vegetables [b]	NA	0.233	0.42	0.42
Fruits [b]	NA	0.116	0.328	0.328
Root Vegetables [b]	NA	0.106	0.173	0.173
Beef [b]	NA	NA	0.485	0.485
Dairy [b]	NA	NA	0.254	0.254
Fish[b]	NA	NA	0.133	NA

a - Parameters Guidance, March 1997
b - Exposure Factors Handbook, 1998. Table 12-17 Fraction of Food Intake that is Home Produced

5.3.1 Child Receptor

This section focuses on the child receptor because they are expected to be a more sensitive subpopulation and EPA recognizes the need to consider health risks to children. EPA has yet to establish policy guidelines that detail the methodology to be used to assess risks to children. The methodology employed in this analysis differentiates risks between adults and humans based solely on different exposure profiles. Toxicity values (i.e., RfD, RfC, and CSF) were not adjusted as these values are developed to be protective of more sensitive subpopulations.

5.3.1.1 Dietary Intakes for Children. This risk analysis provides a preliminary conservative approach for addressing exposure to children. The consumption rates for children of various ages must be keyed to the body weight for that age group. In the EFH, ingestion rates for all dietary items are presented by the following age groups: 1 to 2, 3 to 5, 6 to 11, and 12 to 9 years. The EFH also presents body weight distribution charts for males and females ages 6 to 11 months and in 1-year increments thereafter to age 19. In order to obtain average body weights for use with the dietary intake tables, body weights for both sexes for each year of age are needed. The average body weights provided in EFH are presented in Table 5-7.

Table 5-7. Average Body Weights for Children by Age

	Average Wt. for Age (Kg)																		
Age group	**1 to 2**		**3 to 5**			**6 to 11**						**12 to 19**							
Age/ Sex	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**
male	11.8	13.6	15.7	17.8	19.8	23	25	28.2	31.1	36.4	40.3	44.2	49.9	57.1	61	67.1	66.7	71.1	71.7
female	10.8	13	14.9	17	19.6	22.1	25	27.9	31.9	36.1	41.8	46.4	50.9	54.8	55.1	58.1	59.6	59	60.2

The body weights for both males and females in each year of the age range are averaged to obtain the average body weight for each age group. These average weight values are presented in Table 5-8.

Table 5-8. Average Body Weights for Age Ranges

Age Range (Yrs)	Weights (kg)
1 to 2	12.3
3 to 5	17.5
6 to 11	30.7
12 to 19	58.3

This average body weight (kg) for the age groups is used to estimate the daily consumption rate distributions (kg/d) for the age groups from the intake rate distributions (kg diet/kg body weight -day) presented in the EFH. This approach will yield a distribution of consumption rates for children in each age range. However, children may be any age range during the period of exposure and may not remain in a single age range throughout the selected exposure duration. For example, the child's high-end exposure duration is 18 years. Obviously the child will span all of the aforementioned age ranges during this period of exposure. For long exposure durations, some type of weighted average ingestion rate may be used; however, this may not be appropriate for shorter exposure durations. For this analysis the age range with the highest consumption rate is selected for use. This consumption rate is used for all children for all exposure durations. This is a conservative assumption for exposure durations larger than the age ranges used. For all dietary items except dairy products, the age range with the highest intake is 12 to 19 years. For dairy products, the age range with the highest consumption rate is 6 to 11 years. Options for combining consumption rates and exposure durations for growing children are important for all dietary items. This issue is under continuing development. The child's consumption rates that were used in this analysis are presented in Table 5-4.

5.3.1.1.1 ***Dairy Intake for Children***. Dairy intake rates for children uses a different source than all other intake parameters. All data for the other intake parameters are from the tables of intakes of home-produced food items. The data for dairy intake for children are from the EFH's Appendix 3A, "Sample Calculation of Mean Daily Fat Intake Based on CDC (1994) Data." Table 3-2 in that appendix presents intake of total dairy products by age groups. These data represent per capita intake rates and are not limited to home-produced products. In the absence of other data sources on home produced dairy products, it is recommended that these data be used. The home-produced products section does give a fraction of home-produced dairy items consumed that can be used to develop a distribution to be used in the analysis. The fractions of dietary intakes that are home produced, and thus considered contaminated, are presented in above in Table 5-6.

5.4 Exposure Durations

Data for exposure duration are obtained from the distributions presented for population mobility (Chapter 14.3 of the EFH). There are data for numerous categories of residents. For children, the exposure duration will change from using only the traditional 6 years' exposure duration to using the distribution of values presented in Table 14-159 of the EFH, "Descriptive Statistics for Both Genders by Current Age." These data are relatively constant over childhood and can be adapted for use as a general distribution of values for children up to age 18.
Table 5-9 presents estimated mobility data for children based upon current age.

Table 5-9. Descriptive Statistics for Population Mobility for Children by Current Age

	Residential Occupancy Period (yrs)					
	Percentile					
Current Age (yr)	**25**	**50**	**75**	**90**	**95**	**99**
3	3	5	8	13	17	22
6	4	7	10	15	18	22
9	5	8	12	16	18	22
12	5	9	13	16	18	23
15	5	8	12	16	18	23
18	4	7	11	16	19	23
Average	**4.3**	**7.3**	**11**	**15.3**	**18**	**22.5**

5.5 Results and Discussion

5.5.1 Bounding Human Health Analysis Results

As discussed in Section 2.2.1 the bounding analysis was designed to be extremely conservative. No scenarios bounded out completely; however, most constituents showed no cancer risks greater than 1E-6 or hazard quotients greater than 1.0 for non-carcinogens for most scenarios.[6] The results of this analysis indicated the need for more realistic analyses for some constituents. Since many additional constituents needed to be modeled for the ecological risk analysis, all constituents that were assessed in the bounding analysis were again assessed in the high-end analysis.

5.5.2 High-end Human Health Analysis Results

Result summaries are presented in tables 5-11 through 5-17. These summaries present only WMU/chemical/receptor combinations that had a cancer risk of greater than 1E-6 or a hazard quotient, for non-carcinogens, of greater than 1.0 (carcinogen risk values are presented in scientific notation while non-carcinogen hazard quotients are presented in standard notation). If a WMU/chemical/receptor combination met this criterion, then the highest risk value is reported and the high-end parameter combination that led to this result is presented by code letters below the risk or hazard quotient value in the summary table. Table 5-10 lists the letter codes for the parameters used in the exposure modeling.

A full set of the human health risk analysis results are presented in Appendix H. Similarly, cancer risk numbers are presented in scientific notation and non-cancer hazard quotients are presented in standard notation. All cancer risk or hazard quotient exceedences have a bold border and the number is italicized. The maximum value for that WMU/chemical/receptor combination is in bold font in addition to having a bold border and being italicized.

Note that the results tables present values for beryllium as a carcinogen. The oral CSF for beryllium was withdrawn from IRIS on April 3, 1998. Risk values had already been generated and presented in this document for beryllium using the oral CSF before the CSF was withdrawn. Therefore, ingestion risk values for beryllium are no longer valid. Section 5.8 below addresses the potential for adverse human health effects from beryllium based on its RfD.

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[6] The term“bound out” refers to the a situation when there is no excess risk or hazard quotients. When this occurs, the constituent or waste stream can be omitted from further analysis.

Table 5-10. Parameters Varied in High-end Analysis and Accompanying Codes Used in Results Tables

High-End Code	High-end Parameter[1]	Soil Amendment High-end Parameter
Z	Beef Intake Rate	Beef Intake Rate
A	Dairy Intake Rate	Dairy Intake Rate
B	Above-ground, Exposed Vegetable Intake Rate	Above-ground, Exposed Vegetable Intake Rate
C	Root Vegetable Intake Rate	Root Vegetable Intake Rate
D	Fish Intake Rate	Fish Intake Rate
E	Fruit Intake Rate	Fruit Intake Rate
F	NA	Waste Application Rate
G	NA	Waste Application Frequency
H	Starting Waste Concentration	Starting Waste Concentration
I	NA	Tilling Depth
J	Adult Soil Intake Rate	Adult Soil Intake Rate
K	Child Soil Intake Rate	Child Soil Intake Rate
L	Meteorological Data	NA
M	Distance to Receptor	NA
N	Waste Management Unit Area	NA
Y	Long Exposure Duration	Long Exposure Duration
U	Single Variable Place Holder	Single Variable Place Holder

1 High-end parameters for all scenarios except agricultural soil amendment scenario.

Table 5-11. Draft Human Health Summary Results for Utility Coal Co-managed Wastes Managed in an Onsite Landfill

		Nickel	Silver	Thallium	Arsenic	Barium	Beryllium	Boron	Cadmium	Chromium VI	Cobalt	Selenium
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	2.5E-6 HN	–	–	–	–	–	–	–
	Child of Farmer	–	–	1.1 HN	1.7E-5 HK	1.0 HN,HK	1.3E-5 KN	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	2.3E-6 HY,NY	–	–
	Farmer	–	–	–	–	–	–	–	–	3.5E-6 HN	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	2.5E-6 HN	–	–

Table 5-12. Draft Human Health Summary Results for Utility Coal Co-managed Wastes Managed in a Dewatered Surface Impoundment

		Nickel	Silver	Thallium	Arsenic	Barium	Beryllium	Boron	Cadmium	Chromium VI	Cobalt	Selenium
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	2.1E-6 HL	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	2.0E-6 HL	–	–	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	1.5E-6 HY,LY	–	–
	Farmer	–	–	–	–	–	–	–	–	2.2E-6 HL	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	1.6E-6 HL	–	–

Table 5-13. Draft Human Health Summary Results for Utility Oil Wastes Managed in Onsite Landfill

		Nickel	Silver	Thallium	Arsenic	Barium	Boron	Cadmium	Chromium VI	Cobalt	Vanadium	Zinc
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	5.3E-6 HK	–	–	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	–	–	–

Table 5-14. Draft Human Health Summary Results for FBC Wastes Managed in Onsite Landfill

		Nickel	Silver	Thallium	Arsenic	Barium	Beryllium	Boron	Cadmium	Chromium VI	Cobalt	Vanadium
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	1.0E-6 HY	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	8.2E-6 HK	–	4.7E-6 HK	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	1.0E-6 MY	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	–	–	–

Table 5-15. Draft Human Health Summary Results for FBC Wastes Used as Agricultural Soil Amendment

		Nickel	Silver	Thallium	Arsenic	Barium	Beryllium	Boron	Cadmium	Chromium VI	Cobalt	Vanadium
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	–	–	–

Table 5-16. Draft Human Health Summary Results for Non-utility Coal Co-managed Wastes Managed in an Onsite Landfill

		Nickel	Silver	Thallium	Arsenic	Barium	Beryllium	Boron	Cadmium	Chromium VI	Cobalt	Selenium
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	1.2E-6 HN	–	–	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	–	–	–

Table 5-17. Draft Human Health Summary Results for Non-utility Co-disposed Wastes Managed in an Offsite Commercial Landfill

		Nickel	Silver	Thallium	Arsenic	Barium	Beryllium	Boron	Cadmium	Chromium VI	Cobalt	Vanadium
Ingestion	Adult Resident	–	–	–	–	–	–	–	–	–	–	–
	Home Gardener	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	4.9E-6 HN	–	4.2E-6 KN	–	–	–	–	–
	Fisher	–	–	–	–	–	–	–	–	–	–	–
Inhalation	Adult Res./ Home Gard.	–	–	–	–	–	–	–	–	–	–	–
	Farmer	–	–	–	–	–	–	–	–	–	–	–
	Child of Farmer	–	–	–	–	–	–	–	–	–	–	–

5.6 Discussion of Results

The following subsections discuss the results of the high-end analysis presented above. Each subsection will present the maximum calculated risk or hazard quotient and will also discuss the number of exceedences that were observed. Exceedence is defined as a cancer risk of 1E-6 or greater or a non-cancer hazard quotient (HQ) of 1.0 or greater. Also, the number of risks or hazard quotients that are of potential concern are addressed. Risks or hazard quotients of potential concern are defined as values within one order of magnitude of the risk or HQ level of concern (i.e., risks greater than or equal to 1E-7 or HQ's greater than or equal to 0.1.) Finally, the driving risk parameters for the scenario and pathway under consideration are discussed.

5.6.1 Coal-fired Utility Co-managed Wastes

5.6.1.1 Onsite Landfill Scenario.

5.6.1.1.1 *Ingestion Risks*. For the ingestion pathways for both the farmer and child receptors arsenic had risk exceedences. The maximum risk value for the child was 1.7E-5 and the maximum risk value for the farmer was 2.5E-6. In addition, both barium and thallium had a hazard quotient (HQ) that exceeded the threshold HQ (greater than or equal to 1.0) for the child receptor. The maximum HQ for barium was 1.0 and the maximum HQ for thallium was 1.1.

Including the central tendency run, all single-variable high-end runs, and all double-variable high-end runs, a total of 67 model runs were performed for the farmer receptor. From these 67 model runs the farmer had 5 exceedences for arsenic. All other model runs (model runs that did not result in an exceedence) resulted in arsenic risks are greater than 1E-8 and a significant portion (greater than 1/2) of these risks are greater than 1E-7. The child receptor had a total of 79 model runs and these runs resulted in 52 exceedences for arsenic, 4 exceedences for thallium, and 2 exceedences for barium. All other model runs resulted in arsenic risks greater than 1E-7 and thallium HQ's greater than 0.1 . For barium, 23 of the other model runs resulted in HQ's greater than 0.1.

The starting waste concentration was the driving risk parameter for both the farmer and child receptors -- being one of the high-end parameters in all maximum exceedences. Other parameters that contributed to exceedences for the farmer include: exposure duration, WMU area, dairy intake rate, and distance to receptor. For the child receptor the most important parameters, other than starting waste concentration include: child soil intake rate, distance to receptor, and WMU area.

5.6.1.1.2 *Inhalation Risks*. Chromium VI is the only constituent that showed excess risks for the inhalation pathway. It did so for all receptors. The highest risk value was 3.5E-6 for the farmer. The other maximum risks were 2.5E-6 for the child and 2.3E-6 for the adult resident/home gardener.[7]

Including the central tendency model run, the single-variable high-end runs, and the double-variable high-end runs there were 16 model runs performed. The farmer showed six exceedences for chromium VI while the child showed five exceedences for chromium VI and the adult resident/home gardener showed four exceedences for chromium VI. All results, except for two results for the adult resident/home gardener, showed risk levels greater than 1E-7. There also were several arsenic risk levels that exceeded 1E-7 and a small number of barium HQ's that exceeded 0.1.

Similar to the ingestion pathway, the starting waste concentration is the driving risk parameter -- being one of the high-end parameters in all maximum exceedences. Other important parameters include exposure duration, distance to receptor, and WMU area.

5.6.1.2 Dewatered Surface Impoundment Scenario.

5.6.1.2.1 *Ingestion Risks*. Only arsenic showed exceedences for this exposure pathway. Both the farmer and child receptors showed excess risks with maximum values of 2.1E-6 and 2.0E-6 respectively.

The farmer showed five exceedences in 67 model runs (as discussed in Section 5.6.1.1.1, there were a total of 67 model runs including central tendency, single-variable high-end, and double variable high-end runs) while the child showed three exceedences in 79 model runs. All other arsenic results (results other than the exceedences) for both the farmer and child yielded risks greater than 1E-8 with the majority of the risk values greater than 1E-7. No other constituent showed risks or HQ's within an order of magnitude of the target level.

Starting waste concentration was the driving risk parameter for both the child and farmer receptors. Other important high-end parameters include meteorological location, exposure duration, WMU size, distance to receptor, and dairy products intake rate.

5.6.1.2.2 *Inhalation Risks*. Chromium VI is the only constituent that showed excess risks for the inhalation pathway. It did so for all receptors. The highest risk value was 2.2E-6 for the farmer receptor. The other maximum risks were 1.6E-6 for the child and 1.5E-6 for the adult resident/home gardener.

[7] All adult receptors have the same inhalation rate, exposure time, exposure frequency, body weight, and are exposed to the same air concentration. Therefore, the only difference between receptors is exposure duration. Because the home gardener and adult resident have the same exposure durations, they are combined for the inhalation pathway.

The farmer scenario showed six exceedences out of 16 model runs for Chromium VI while the child showed only one exceedences and the adult resident/home gardener showed two exceedences. All results, except for two results for the adult resident/home gardener receptor and the central tendency results for all receptors, showed risk levels greater than 1E-7. There also are several arsenic risk levels that exceed 1E-7 for all receptors but there are no exceedences for arsenic.

Starting waste concentration was the driving risk parameter -- being one of the high-end parameters in all maximum exceedences. Other driving high-end parameters include exposure duration, meteorological location, and distance to receptor.

5.6.2 Oil-fired Utility Wastes

5.6.2.1 Onsite Landfill Scenario.

5.6.2.1.1 *Ingestion Risks*. Only arsenic for the child receptor resulted in excess risks with a maximum risk of 5.3E-6. Eleven of 67 model runs showed excess risks for the child receptor. (Note that there were only 67 model runs for the child receptor for this scenario because there is no high-end WMU size.) Four other child model runs resulted in risks greater than 1E-7. For the farmer, there were no risks greater than 1E-6; however, 10 runs resulted in arsenic risks greater than 1E-7. For both the home gardener and the adult resident, one run resulted in arsenic risks greater than 1E-7. In addition, all vanadium HQ's were greater than 0.1 for the child receptor.

Starting waste concentration was the driving risk parameter showing an excess risk when modeled as the single high-end parameter and therefore showing risks when combined with all other high-end parameters. No other high-end parameter was significant in itself.

5.6.2.1.2 *Inhalation Risks*. There were no exceedences for the inhalation exposure pathway. There were however several chromium VI risk results that were greater than 1E-7 for all receptors (seven exceedences for both the child and farmer and three exceedences for the adult resident/home gardener). In addition, there were a few nickel risk values that were greater than 1E-7 for all receptors (one for the adult resident/home gardener, three for the farmer, and two for the child) and one arsenic value greater than 1E-7 for both the farmer and child receptor.

5.6.3 FBC Wastes

5.6.3.1 Onsite Landfill Scenario.

5.6.3.1.1 *Ingestion Risks*. Only arsenic showed exceedences for this pathway. Both the farmer and child showed excess risks with maximum risk values of 1.0E-6 and 8.2E-6 respectively.

The farmer showed only one exceedence in 67 model runs while the child showed 22 exceedences in 79 model runs. All other arsenic results for both the farmer and child yielded risks greater than 1E-8 with the majority of the risks greater than 1E-7. In addition, the adult resident and home gardener each yielded two results with risks greater than 1E-7 for arsenic. There also were 11 instances where vanadium yielded HQ's greater than 0.1 for the child receptor.

Waste concentration was the driving risk parameter for both the farmer and child receptors. Child soil intake was also an important risk parameter for the child receptor.

5.6.3.1.2 ***Inhalation Risks*****.** There was only one exceedence for this exposure pathway — one model run for the farmer yielded a risk of 1.0E-6. All but two runs yielded risks greater than 1E-7 for the farmer receptor and a significant portion of the runs for the adult resident/home gardener and child receptors yielded risks greater than 1E-7 (seven for the adult resident/home gardener and 10 for the child).

Distance to receptor and exposure duration were the driving risk parameters.

5.6.3.2 **<u>Agricultural Soil Amendment Scenario</u>.** There were no exceedences for this scenario. Further, there were no risks in excess of 1E-7 nor any HQ's greater than 0.1.

5.6.4 Coal-fired Non-utility Wastes

5.6.4.1 **<u>Onsite Landfill Scenario</u>.**

5.6.4.1.1 ***Ingestion Risks*****.** There was only one exceedence for this pathway. Arsenic for the child receptor yielded a risk value of 1.2E-6 when both the waste concentration and WMU area were set at high-end. Twenty-two of the other runs for the child yielded results greater than 1E-7. Only one run for the farmer scenario yielded a risk value greater than 1E-7. There was only one occurrence of an HQ greater than or equal to 0.1. This was for thallium for the child receptor with a value of 0.1.

5.6.4.1.2 ***Inhalation Risks*****.** There were no inhalation risk exceedences for this pathway. Further, there were only three occurrences (two for the farmer and one for the child) of risk values greater than 1E-7 -- the greatest of these values being 1.4E-7.

5.6.4.2 **<u>Offsite Commercial Landfill Scenario</u>.**

5.6.4.2.1 ***Ingestion Risks*****.** Only arsenic ingestion for the child receptor showed excess risks for this scenario. The maximum child risk was 4.9E-6.

The child showed four exceedences in 79 model runs. Slightly more than ½ of the remaining child risks were greater than 1E-7. The child receptor also had 11 HQ's for thallium greater than 0.1 and two HQ's for barium greater than 0.1. Also, the farmer receptor showed six risk results for arsenic greater than 1E-7.

Waste concentration was the driving risk parameter. Other important parameters included WMU area, child soil ingestion rate, and distance to receptor.

5.6.4.2.2 *Inhalation Risks*. There were no inhalation risk exceedences for this scenario. For chromium VI there were occurrences of risk greater than 1E-7. These included one occurrence for the adult resident, four occurrences for the farmer, and two occurrences for the child with the highest risk being 2.2E-7 for the farmer.

5.7 Risks from Mercury

Modeling mercury in the environment is a complex issue. Mercury was not modeled in this analysis. Mercury levels are low in the wastes in which it was detected. There was no mercury reported in the coal co-managed waste analytical data. For FBC waste the maximum concentration was 2.78 ppm while the maximum concentration in oil combustion wastes was 0.38 ppm. Due to the volatility of mercury and the environment of the fossil fuel combustion process, most mercury is emitted from the stack.

The major reason for non-inclusion in this analysis is that the risk assessment methodology for mercury is much more complex than for other metal constituents and the methodology is under review by EPA's Office of Research and Development. Fate and transport modeling for mercury requires a complex analytical framework that accounts for both the species-specific mobility of mercury between media compartments as well as the interconversion between different species as mercury equilibrates within individual compartments. A draft Report to Congress (RTC) for Science Advisory Board (SAB) review was released in June 1996. This RTC proposed a methodology for modeling mercury. Based on SAB review, a final RTC was issued in January, 1998 that included significant changes to the methodology presented in the review draft RTC. The timeframe for this analysis did not allow for the new methodology to be developed into a modeling tool and utilized in this project.

Finally, even if the modeling methodology and capabilities were developed, speciation data at the source are needed to employ the methodology in the RTC. Only total mercury concentrations were supplied in the analytical data provided to RTI.

5.8 Risks from Lead

Because lead does not have human health toxicity benchmarks (i.e.,RfC, RfD, CSF) risks or hazard quotients cannot be calculated. However, lead does have the potential for adverse health and developmental effects especially for children. To estimate the potential for adverse effects from lead, maximum lead soil concentrations for each scenario were compared to the lead screening level of 400 mg/kg from *Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities* (U.S. EPA, 1994). The *Risk Assessment Guidance for Superfund* (RAGS) (U.S. EPA, 1989) has established this is the maximum safe lead concentration for child soil ingestion based on a risk analysis. Table 5-18 presents the maximum lead soil concentrations that were calculated for each scenario that was modeled. All lead soil concentrations fall well below 400 mg/kg. It is therefore concluded that lead soil levels resulting from the management and use of FFC wastes are unlikely to cause a significant threat to human health.

5.9 Risks from Beryllium

The ingestion CSF for beryllium was removed from the Integrated Risk Information System (IRIS) on April 3, 1998. All modeling for this project had already been completed using the ingestion CSF of 4.3 $(mg/kg/day)^{-1}$. The RfD for beryllium has also recently been revised to 2E-3 mg/kg/day. To ascertain the potential for adverse health effects from beryllium utilizing the new RfD from IRIS the same scenario that resulted in the highest cancer risk value for beryllium was duplicated except the RfD was utilized instead of the CSF. This would result in the maximum HQ that would be expected to occur for all scenario/pathway/receptor combinations. Therefore, the child ingestion pathway for the utility coal-fired co-managed waste managed in an onsite landfill scenario with child soil ingestion and WMU area set to high-end values was recalculated using the RfD instead of the CSF. The resulting HQ was 0.02. Therefore, it is assumed that beryllium in FFC wastes, as modeled, does not pose significant threat to human health.

Table 5-18. Maximum Estimated Lead Concentrations in Soil Per Scenario

Scenario	Lead Soil Concentration (mg/kg)
Utility Coal-fired Co-managed Waste Onsite Landfill	7.63
Utility Coal-fired Co-managed Waste Dewatered Surface Impoundment	1.30
Utility Oil-fired Waste Onsite Landfill	0.98
FBC Onsite Landfill	0.81
FBC Used as Soil Amendment	0.009
Non-utility Coal-fired Waste Onsite Landfill	0.21
Non-utility Coal-fired Waste Offsite Landfill	1.33

6.0 ECOLOGICAL RISK ASSESSMENT OF COMANAGED FOSSIL FUEL COMBUSTION RESIDUALS

This section describes the ecological risk assessment (ERA) developed to evaluate the potential ecological risks associated with the management and use of comanaged fossil fuel combustion (FFC) residuals. This section is intended to complement the human health risk analysis and the SAIC Background Document and, therefore, detailed information on management practices and fate and transport modeling of FFC constituents has not been included. This section is organized as follows. Section 6.1 describes the overall technical approach for the ERA process of comanaged FFC residuals. Sections 6.2 through 6.4 describe the three basic phases of the ERA process: problem formulation; analysis; and risk characterization; and provide sufficient detail to repeat all calculations used in the analysis. Two critical products from this ERA are included in Appendices I and J, respectively: (1) the stressor-response profiles for constituents of potential ecological concern (CPECs) found in FFC residuals and (2) the calculation spreadsheet used to estimate the ecological risks associated with the management and use of FFC residuals. The calculation spreadsheet includes all of the inputs required to calculate chemical stressor concentration limits (CSCLs) for the CPECs considered in this analysis, background soil concentrations of CPECs, and comparisons of CSCLs developed for the FFC ERA with criteria used in other programs and agencies. Conclusions are summarized in Section 6.5

6.1 Technical Approach for ERA of Comanaged FFC Residuals

The technical approach developed to evaluate the potential for adverse ecological effects associated with the management and use of comanaged FFC residuals follows EPA's Guidelines for Ecological Risk Assessment (U.S. EPA, 1996a). The *Guidelines* describe the three basic phases that frame the ecological risk assessment process, namely; problem formulation, analysis, and risk characterization. The framework advocated in the *Guidelines* has been adopted by OSW for all ecological risk assessment work conducted in support of regulatory determinations (e.g., HWIR, silver delisting, hazardous waste combustion). Briefly, these phases may be summarized as follows:

Problem formulation phase - Defines the problem by answering these questions: (1) What are the constituents of concern? (2) Once released, what is the environmental behavior of the constituents (e.g., persistence, bioaccumulation, speciation)? (3) Given the source characterization (e.g., size, geographical location), what ecosystems and ecological receptors are potentially at risk? (4) What adverse ecological effects are possible following exposure? The three key activities in this phase are: (1) the selection of assessment endpoints, (2) development of a conceptual model, and (3) preparation of an analysis plan.

Analysis phase - Provides estimates of the constituent concentrations in the environment to which ecological receptors are exposed (i.e., exposure profile) and develops chemical stressor concentration limits, or CSCLs, from data on adverse ecological effects on various receptors (i.e., acceptable concentrations in environmental media).

Risk characterization phase - Compares the modeled exposure concentrations to the chemical stressor concentration limits (CSCLs) developed in the analysis phase to estimate the potential for adverse ecological effects (i.e., the hazard quotient, or HQ, approach). Includes a risk description of the assessment (e.g., limitations) and discusses the ecological significance of HQ exceedances.

The technical approach shown in Figure 6-1 was developed to provide a tiered strategy for the assessment of ecological risks. The tiers progressed from a highly conservative on-site screening assessment to an off-site modeling exercise of CPEC release and transport for central tendency management and use practices. The tiers consisted of four different assessments for management and use practices of comanaged FFC residuals:

1. on-site screening (problem formulation);
2. bounding assessment (analysis phase);
3. high-end assessment (analysis phase); and
4. central tendency assessment (analysis phase).

Although the progression of the bounding assessment through the central tendency assessment is shown in the analysis phase, the process was iterative in that CPECs could be removed from further consideration at any point in the progression. Hence, the decision in Item #7 (Do the modeled exposure concentrations exceed the CSCLs?) could have resulted in removal of the CPEC for the bounding, high-end, or central tendency assessments. Each tier is described briefly below.

On-site screening - As part of the problem formulation, CSCLs were compared directly with maximum concentrations of CPECs found in comanaged FFC residuals (both total concentrations and pore water concentrations). Although direct comparison of ecological effects levels with waste concentrations is highly conservative, the results provided important insights into the potential ecological risks and were instructive in identifying CPECs . For example, concentrations of a number of CPECs were below the CSCLs or were within a factor of 10. Given the level of conservatism in the comparison, the CPEC/CSCL ratios below 10 were considered to be strongly suggestive of *de minimis* potential ecological impacts.

Figure 6-1. Technical Approach Developed for Era of Management & Use of Comanaged Ffc Residuals

Bounding assessment - During the analysis phase, all parameters for the fate and transport model (including source area sizes and concentrations) were set at high-end values that approximated the 95th percentile. The off-site concentrations in soils and surface waters represent the extreme tail of the exposure distribution and, therefore, comparison with the CSCLs provided a conservative tool to evaluate off-site exposures. It is important to recognize that the intent of the bounding assessment is to indicate whether additional ecological analyses are required, not to support a regulatory decision. As with the on-site screening, CPEC/CSCL ratios below 10 were considered to be strongly suggestive of *de minimis* potential for ecological impacts.

High-end assessment - During the analysis phase, CPECs that were not ruled out for the bounding analysis were evaluated for high-end management and use practices. The high-end exposure assessment is defined by setting the two most sensitive model parameters to high-end values (i.e., approximating 95th percentile) and setting all other model parameters to central tendency values. In virtually all cases, the two parameters set to high-end values were source concentration and source area.

Central tendency assessment - During the analysis phase, CPECs that were not ruled out for the high-end assessment were evaluated for central tendency management and use practices. The central tendency assessment is defined by setting all model inputs to central tendency values. Note that the CSCLs and underlying assumptions (e.g., 100% of the diet originates on contaminated area) remain constant throughout the analysis. Consequently, the comparison of CSCLs with off-site CPEC concentrations is, for most ecological receptors, inherently conservative.

6.2. Problem Formulation

As described in EPA's Proposed Guidelines for Ecological Risk Assessment (U.S. EPA, 1996), a successful problem formulation "depends upon the quality of three products: (1) assessment endpoints that adequately reflect management goals and the ecosystem they represent, (2) conceptual models that describe key relationships between a stressor and assessment endpoint or among several stressors and assessment endpoints, and (3) an analysis plan." Although these products are presented sequentially below, it is important to note that the problem formulation phase (like the ERA process) is iterative in nature.

6.2.1 Selection of Assessment Endpoints

Perhaps the most important step in the problem formulation phase is the selection of assessment endpoints, defined as "explicit expressions of the actual environmental value that is to be protected" (EPA, 1996). The assessment endpoints serve as a critical link between the ecological risk assessment (ERA) and the management goal which, for the FFC ERA, may be stated as follows: "to evaluate the potential for adverse ecological effects associated with the management, and/or use of comanaged FFC residuals." The assessment endpoints must be ecologically relevant to the ecosystem(s) they represent and susceptible to the stressors of concern, in this case the CPECs found in FFC residuals. Candidates for assessment endpoints often include threatened/endangered species, critical habitats and ecosystems, commercially or recreationally important species, functional attributes that support food sources or flood control, or aesthetic values, such as the existence of charismatic species like eagles (U.S. EPA 1996a). Regardless of the assessment endpoint(s) chosen for this analysis, it should be emphasized that each assessment endpoint is defined by two key elements: (1) a valued ecological entity (e.g., a species) and (2) an attribute of that entity is important to protect (e.g., reproductive fitness).

For the FFC ERA, the **assessment endpoints** were chosen to be consistent with the assessment endpoints selected for the proposed Hazardous Waste Identification Rule (HWIR). As with the HWIR, ecological exposures are presumed to occur anywhere in the contiguous United States. Consequently, a suite of assessment endpoints was chosen based on: (1) their significance to the ecosystem, (2) their position along a continuum of trophic levels, and (3) their susceptibility to metals based on exposure and/or toxicological sensitivity. In Table 6-1, the assessment endpoints (i.e., values to be protected) are defined in terms of: (1) the significance of an ecological entity, (2) the ecological receptor representing that entity, (3) the characteristic about the entity that is important to protect, and (4) the measures of effect used to predict risk. The intent of including multiple receptors is that, by protecting producers (i.e., plants) and consumers (i.e., predators) at different trophic levels, as well as certain structural components (e.g., benthic community), a degree of protection from CPECs may be inferred to the ecosystem as a whole.

In addition to evaluating representative species populations and communities, it is also important to consider the potential effects on, managed lands (e.g., National Wildlife Refuges), critical habitats (e.g., wetlands), and threatened and endangered species. Although metrics to evaluate the impacts on the ecological "health" of these entities are not available for use in this type of analysis, the presence of valued habitats and species may require a different approach in assessing ecological impacts associated with CPEC releases. However, these assessment endpoints were not evaluated since the FFC ERA was a national rather than a site-based analysis.

Subsequent to assessment endpoint selection, the **conceptual model** for this bounding analysis was developed through a synthesis of information generated under the activities indicated in Item #3 in Figure 6-1. By integrating the data developed under each activity, a conceptual model was developed that included three source scenarios: (1) landfilling of FFC residuals, (2) management in surface impoundments (i.e., settling basins), and (3) application to agricultural soils. For each scenario, potential ecological receptors were identified along with exposure pathways of concern. This conceptual model frames the entire screening analysis of ecological risks and, therefore, each of these activities is described below in Sections 6.2.1.1 through 6.2.1.4. Some sections are intentionally brief to avoid duplication with the SAIC Background Document.

6.2.1.1 Characterization of Constituent Behavior and Toxicity

Constituent behavior - The CPECs in comanaged FFC residuals include essential and non-essential metals at concentrations both above and below background soil concentrations in the United States (see Table 2.2 in Appendix J for a list of soil background concentrations). Nevertheless, the behavior of metals in the environment depends on the specific geochemistry of the ecosystem as defined by a number of characteristics such as: pH, concentration of competing constituents (e.g., other metals), availability of inorganic moieties to react with metal (e.g., sulfide and chloride ions), concentration of dissolved organic matter (DOC), and concentration of total suspended solids (TSS). For example, studies on fish have shown that organic acids (e.g., humic, fulvic) bind tightly to metals reducing their bioavailability and, ultimately, their toxicity (Hutchinson and Sprague, 1987). Thus, water quality characteristics directly influence the partitioning of metals between particles and soluble species and determine the fraction of free ionic metal to which biota are exposed. This last point is particularly important for aquatic biota since a preponderance of studies have demonstrated that organism response is correlated with free metal ion (Allen and Hansen, 1996). Section II of the stressor-response profile in Appendix I of this report contains a concise discussion of the geochemistry of each CPEC in soils, surface waters, and sediments, respectively. The profiles pay particular attention to the predominant metal species in each medium as well as the relationship between bioavailability and environmental characteristics in terrestrial and aquatic ecosystems (e.g., pH, DOC).

Constituent toxicity - The majority of the ecotoxicological data identified for CPECs was based on acute and chronic laboratory results for mammals, birds, terrestrial plants, aquatic species, (e.g., fish, invertebrates, algae, amphibians, and the benthic community), and the soil community (e.g., earthworms). Ranges of effects are presented in Section III of the stressor-response profiles (see Appendix I). In addition, the profiles address the following issues:

- concentrations for acute and chronic endpoints;
- potential for bioaccumulation;
- conditions that influence bioavailability;
- relative sensitivities of various receptors; and
- ecotoxicity data gaps.

A figure is provided in each profile illustrating the range of acute and chronic effects data in general taxa groups. Information regarding the observed responses of exposed receptors (e.g., toxicity endpoints such as organ damage, teratogenecity, neurotoxicity) are outlined briefly.

Table 6-1. Suite of Assessment Endpoints Considered for the FFC ERA

Ecological Significance	Assessment Endpoint	Receptors	Characteristic(s)	Measure of Effect
• Upper trophic level consumers • Socially valued (e.g., endangered species) • Top recipients of bioaccumulative chemicals • Represent species with large foraging ranges • Represent species with longer life spans	viable mammalian wildlife populations	e.g., deer mouse, meadow vole, red fox	reproductive and developmental success	chronic or subchronic NOAEL(s) or LOAEL(s) for developmental and reproductive effects
	viable avian wildlife populations	e.g., red-tailed hawk, northern bobwhite	reproductive and developmental success	chronic or subchronic NOAEL(s) or LOAEL(s) for developmental and reproductive effects
• Species represent unique habitat niches (e.g., partially aquatic and terrestrial) • Some species are sensitive to contaminant exposure	viable amphibian and reptile wildlife populations ("herps")	e.g., frog, newt, snake, turtle	reproductive and developmental success	chronic or subchronic NOAEL(s) or LOAEL(s) for developmental and reproductive effects
• Represents base food web in terrestrial systems • Habitat vital to decomposers and soil aerators • Proper soil community function related to nutrient cycling	sustainable soil community structure and function	e.g., nematodes, soils mites, springtails, annelids, arthropods	growth, survival, and reproductive success	95% of species below no effects concentration at 50th percentile confidence interval
• Primary producers of energy in ecosystems • Act as food base for herbivores • Able to sequester some contaminants • Can act as vectors to bioaccumulation • Constitute a large fraction of the earth's biomass	maintain primary terrestrial producers (plant community)	e.g., soy beans, alfalfa, rye grass	growth, yield, germination	10th percentile from LOEC data distribution
• Highly exposed receptors from constant contact with contaminated media • Act as vectors to transfer contaminants to terrestrial species	sustainable aquatic community structure and function	e.g., fish (salmonids), aquatic invertebrates (daphnids)	growth, survival, reproductive success	ambient water quality criteria (NAWQC) for aquatic life (95% species protection)
• Provide habitat for reproductive lifestages (e.g., eggs, larval forms) • Habitat for key invertebrate species • Act to process nutrients and decompose organic matter	sustainable benthic community structure and function	e.g., protozoa, flat worms, ostracods	growth, survival, reproductive success	10th percentile from LOEC data distribution
• Primary producers of energy in the aquatic system. • Base food source in the aquatic system • Can act to sequester contaminants from the water column • Act as substrate for other organisms in the water column (e.g., periphyton)	maintain primary aquatic producers (algal & plant community)	e.g., algae and vascular aquatic plants	growth, mortality, biomass, root length	EC_{20} for algae; lowest LOEC for aquatic plants

In addition to laboratory toxicity data, literature describing field studies of the environmental effects of FFC residuals and CPECs was examined. A number of articles were identified linking adverse effects in amphibians (and other biota) with exposure to sluiced ash that is pumped into a series of settling basins and, ultimately, into a two hectare drainage swamp. The drainage swamp at the Savannah River Site historically received effluent from other coal ash settling basins that are no longer in use (Rowe et al., 1996). Various studies related to this site (e.g., Cherry et al., 1984; Guthrie and Cherry, 1979; Hopkins et al., 1997) have demonstrated that effects ranging from mortality to sublethal endpoints such developmental anomalies are causally linked to the presence of elevated concentrations of trace elements found in coal ash residuals. In particular, selenium has been shown to be toxic to amphibians at relatively low levels of exposure (e.g., Browne and Dumont, 1979). In addition, accumulation of selenium (a common trace metal in FFC residuals) has been demonstrated in both terrestrial (e.g., Carlson and Adriano, 1993) and aquatic ecosystems (e.g., Lemly, 1985; Ohlendorf et al., 1990; Saiki et al., 1993). Many studies suggested that effects were caused by the properties of FFC ash (e.g., low pH; siltation; high soluble salts concentration; pozzolanic action) as well as the trace metals in combustion ash.

6.2.1.2 Source Characterization for CPEC Releases from Comanaged FFC Residuals

The source characterization for the management, disposal, and beneficial use of comanaged FFC residuals is described in detail in the SAIC Background Document. In brief, three major types of wastes were evaluated: (1) utility and non-utility comanaged residuals, (2) utility oil combustion residuals, and (3) fluidized bed combustion residuals. As depicted in Table 6-2, the FFC ERA considered potential release and exposure for landfill disposal, surface impoundment management, and application to agricultural areas as a soil amendment (FBC ash only).

Table 6-2. Matrix Summarizing Waste Residuals Evaluated for Each Source

management unit	coal comanaged residuals	FBC combined ash	oil fly ash	oil bottom ash	oil settling basin solids
landfill	**yes**	**yes**	**yes**	**yes**	**yes**
surface impoundment	**yes**	**yes**			
soil amendment		**yes**			

CPECs Selected for FFC ERA
Aluminum
Antimony
Arsenic
Barium
Beryllium
Boron
Cadmium
Chromium
Cobalt
Copper
Lead
Mercury
Molybdenum
Nickel
Selenium
Silver
Thallium
Vanadium
Zinc

Combustion of coal and other fossil fuels produces a variety of residuals including, for example, fly ash, bottom ash, flue gas desulfurization (FGD sludge), fluidized bed combustion (FBC) ash, coal gasification ash, oil fly ash, oil bottom ash, and oil settling basin solids. The physical, chemical, and mineralogical characteristics of these comanaged residuals depend on a variety of factors such as the composition of the parent fuel (e.g., sulfur content in coal), the efficiency and type of emission control devices, the character of comanaged wastes, and the management methods used. As a result, it is difficult to generalize about the composition and properties of FFC residuals or their behavior in the environment (Carlson and Adriano, 1993). For example, ash pH can vary from 4.5 to 12 depending on the S content of the parent coal, with high-S, eastern coals typically producing acidic ashes and low-S, western coals frequently producing alkaline ashes (Adriano et al., 1980). The pH of the ash is particularly important with regard to ecological effects since many metals are more mobile and toxic at lower pHs (e.g., Jung and Jagoe, 1995). Although additional detail on comanaged FFC residuals is available in the SAIC Background Document, it is useful to point out several common characteristics in FFC residuals that are relevant to environmental behavior and potential effects associated with their release:

- many residuals exhibit pozzolanic properties, i.e., they react with water in the presence of lime to form cement (e.g., Adriano et al., 1980);

- most residuals are enriched in trace elements including both essential (e.g, Se, Ca, Zn) and non-essential metals (e.g., As, Hg, Sb);

- trace elements tend to be concentrated on smaller ash particles (Adriano et al., 1980); and

- particle sizes range from 0.01 µm for fly ash to >2 mm for bottom ash comanaged with boiler slag (Carlson and Adriano, 1991).

The constituents of potential ecological concern were selected based on: (1) availability of ecotoxicological data and criteria on metals, particularly those evaluated under the HWIR, (2) evidence of adverse effects and uptake of metals in the scientific literature, especially from case studies involving FFC residuals, and (3) presence of constituents in sampled residuals. Where possible, metal species were considered along with total metal concentrations (e.g., Cr^{3+} and Cr^{6+}; As^{3+} and As^{5+}). However, quantitative criteria were available only for aquatic exposures.

6.2.1.3 Identification of Ecosystems/Ecological Receptors Potentially at Risk

Essentially three types of information were used to identify ecosystems and receptors potentially at risk from exposure to CPECs released from comanaged FFC residuals: source location, CPEC behavior, and ecological effects.

Location of Sources—Because FFC landfills and surface impoundments may be located virtually anywhere within the conterminous United States, ecosystems potentially at risk include the full array of ecoregions as defined by Bailey (1996). Consequently, ecosystems potentially at risk include the same subset as those evaluated for the HWIR proposed rule. In addition, the literature reviewed regarding field assessments of FFC residual management and use practices indicated that impacts were possible on a variety of terrestrial and aquatic habitats (see, for example, Carlson and Adriano, 1993; Cherry et al., 1984; Golden, 1983). As with the HWIR, a generic scheme of freshwater aquatic and terrestrial ecosystems was chosen to provide the screening-level context for selection of ecological receptors. Freshwater ecosystems include a variety of waterbodies such as lakes and streams, and terrestrial ecosystems are soil-based ecosystems such as forests and grasslands. This coarse level of resolution was deemed appropriate for a bounding analysis, particularly since site-specific risk assessment work was not envisioned for this study.

Environmental Behavior of CPECs—Although the geochemistry of the ecosystem will determine what fraction of metal is bioavailable to various biota, the range of partition coefficients indicates that many of the CPECs will be particle-bound and tend to move with the soil particles and deposit to adjacent soils or to nearby surface waters. Indeed, soils and sediments serve as environmental sinks for metal contamination; high sediment levels of trace metals have been shown in an ash drainage system (Guthrie and Cherry, 1979). Consequently, ecological receptors that live in close contact with the contaminated media (soil, sediment) and receptors in the lower trophic levels that are exposed through direct contact and ingestion of contaminated media and prey (e.g., worms) were considered to be potentially at risk. In addition, data on the bioaccumulation potential strongly suggested that food web exposure to upper trophic level predators is unlikely for most of the constituents evaluated in this screen; notable exceptions include: *arsenic* - uptake by plants may result in elevated exposures to herbivores; *mercury* - uptake by aquatic organisms may lead to significant exposures in animals that eat fish and/or aquatic invertebrates; and *selenium* - accumulation from coal ash residuals has been demonstrated in both terrestrial and aquatic systems (e.g., Cherry and Guthrie, 1977; Lemly, 1985; Mandisoza et al., 1980).

Ecological Effects Data on CPECs—As noted above, effects data on trace metals found in comanaged FFC residuals were identified on virtually all types of receptors that are typical of terrestrial and aquatic habitats. Of particular concern are the recent data on trace metal impacts on amphibians (e.g., two articles by Rowe et al., in press in Physiological Zoology and Copiea, respectively). In addition, research conducted by Freda (1991) on amphibians strongly suggested that temporary breeding ponds may be particularly at risk from wastes that are acidic and contain trace metals. As an example, the author noted that a by-product of coal mining is the mobilization of iron pyrite (FeS). This compound is oxidized in the presence of bacteria to form sulfuric acid and, as a result, may enhance the toxicity of trace metals. Importantly, Freda pointed out that an estimated 30% to 50% of all species of caudates (salamanders) and anurans (frogs), respectively, use temporary ponds for breeding and many of these species reproduce in them exclusively. Given the total area likely to be impacted by comanaged FFC residuals, it appears likely that some fraction of these temporary ponds may be affected. It should be noted that the effects of concern from these exposures are often sublethal endpoints that influence the growth, metamorphosis, and subsequent reproduction. Rowe et al. (1996) observed a considerable abundance of bullfrog tadpoles in the settling basins and receiving swamp at the Savannah River Site despite the presence of elevated trace metals, suggesting that conditions were not severe enough to cause widespread mortality in developing larvae of this species. Although visual inspection of the site would not have suggested that any adverse effects were occurring, the study data indicate that oral deformities were occurring in tadpoles in the drainage system and that these deformities result in a reduction in survival ability.

6.2.2.4 Identification of Exposure Pathways of Concern

The identification of exposure pathways of concern is a function of all of the activities conducted during the conceptual model development. These activities share a common goal in that they are intended to characterize the exposure scenario that is evaluated for the on-site screening and the bounding, high-end, and central tendency assessments. Tables 6-3 through 6-5 present the exposure scenarios developed and summarize information on the industrial sector, overall strategy, *exposure pathways*, *routes and receptors*, and key assumptions, issues, and inputs. Ecological exposure pathways considered in the FFC ERA include:

- overland transport and ingestion of contaminated medium and food;
- overland transport and direct contact with contaminated medium;
- aerial dispersion, deposition, and direct contact with contaminated medium;
- aerial dispersion, deposition, and ingestion of contaminated medium and food;
- ingestion of contaminated aquatic organisms from the surface impoundment;
- ingestion of contaminated surface water in the impoundment (i.e., surface impoundment used as drinking water source);
- direct contact with surface impoundment waters for amphibians (acute effects only); and
- leaching to groundwater and subsequent discharge to surface waters and wetlands has not yet been evaluated.

Table 6-3. Development of Exposure Scenario for the ERA of FFC Residuals Comanaged in Landfills

Sectors: comanaged wastes from Utility Coal, Utility Oil, FBC (CIBO)
Waste Concentrations: coal ash residuals (fly ash, bottom ash, FGD sludge), FBC ash, oil fly ash, oil bottom ash, oil settling basin solids
Waste Management: landfill

Strategy: As with the exposure scenario constructed to evaluate health risks, the combination of assumptions (e.g., area, concentrations) tends to overstate the constituent releases to surface pathways and, to a lesser degree, subsurface pathways. Because ecological receptors may be exposed to constituents through the same media via similar pathways as are human receptors, this scenario is also presumed to rule out the following scenarios for ecological risks: (1) coal ash comanagement landfills, (2) dewatered coal ash comanagement impoundments, (3) coal ash minefills, (4) FBC ash landfills, (5) dewatered FBC ash impoundments, (6) FBC ash minefills, and (7) oil ash landfills.

Pathway	Route	Receptors	Key assumptions, issues, and inputs
leaching, recharge to aquatic ecosystem, wetlands	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds, herps	The issue of surface water recharge for metals presents a number of complicated, albeit solvable, modeling issues (e.g., speciation in the subsurface; sediment sorption). Although many metals are relatively immobile in the subsurface due to strong binding attractions to organics and charged ions (e.g., clay), metal mobility is strongly influenced by site-specific soil and aquifer conditions, including pH and binding capacity (e.g., as a function of anion concentrations such as FeOx). Because subsurface movement of these constituents tends to be somewhat limited, exposures to animals that feed on aquatic biota (e.g., piscivores) are presumed to be below exposures for pathways involving overland transport or aerial emissions from a landfill to an aquatic ecosystem (see below). Consequently, food chain exposures may not need to be considered in this pathway. Nevertheless, this scenario may pose ecological risks to receptors in direct contact with contaminated sediments and surface waters and should be evaluated in the future.
overland transport to terrestrial ecosystem	direct contact ingestion	soil biota, plants mammals, birds, herps	Overland transport of metal constituents from a landfill to an adjacent terrestrial ecosystem, through the watershed, and into surface water requires essentially the same assumptions as applied to the health risk scenario. In addition, it is important to note that contamination of an adjacent ecosystem would generally not occur through sheet flow, but rather through rill and channel flow (because the terrestrial ecosystem is not tilled or mixed). As a result, this scenario tends to present a conservative exposure profile for ecological receptors in a terrestrial ecosystem; the pattern of contamination that may actually be observed would likely be a ribboned surface drainage pattern. Any effects on soils would likely be highly localized (i.e., hot spots of metal contamination) and, consequently, the ecological significance to the ecosystem may be low. Moreover, many metal constituents are required for proper growth and maintenance of plants and animals, so it is critical to evaluate the relationship between background concentrations modeled concentrations of metals.

Table 6-3. Development of Exposure Scenario for the ERA of FFC Residuals Comanaged in Landfills

Pathway	Route	Receptors	Key assumptions, issues, and inputs
overland transport in watershed to aquatic ecosystem	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds, herps	Overland transport of metal constituents from a landfill through the watershed and into surface waters and/or wetlands may result in adverse ecological effects in the affected stream reach, pond, or wetland area. For wetlands, in particular, these effects may result in significant ecological impacts to species that rely on the wetland as a nursery or as migratory stopover (e.g., cranes). Metals deposition to sediments may, over time, compromise the structure and function of the sediment community and ultimately impact the overall aquatic community. For example, release data associated with a nine acre fly ash pile environmental restoration site on the Oak Ridge K-25 site indicated that runoff and leachate resulted in low pH in surrounding surface water and a storm drain characterization showed high levels of aluminum, arsenic, and iron sulfate (http://www.ornl.gov/er/k-er-47/c-50.htm).
emissions, dispersion, deposition to terrestrial ecosystem	direct contact ingestion	soil biota, plants mammals, birds, herps	The pattern of contamination associated with aerial emissions of metal constituents from the landfill is much more likely to resemble the homogenous mixing represented in the bounding screen. Metal constituents that accumulate over time in the surficial soil may adversely effect species in the soil community, potentially disrupting critical functions that the community performs (e.g., nutrient cycling). In addition, metals are phytototoxic at levels not far above those required for normal growth and have been shown to be toxic to sessile animals exposed via incidental ingestion of soil or, for bioaccumulative metals, through the food chain. These considerations notwithstanding, the spatial extent of contamination (and resulting effect) is likely to be relatively small and probably of limited ecological significance.
emissions, dispersion, deposition to aquatic ecosystem, wetlands	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds, herps	The contribution to surface water loads via this pathway will be significant only for certain conditions, as described in the development of exposure scenarios for human health. Sediments serve as a sink for metal contaminants and, as with the overland transport pathway, the benthos and aquatic biota are potentially at risk.

Table 6-4. Development of Exposure Scenario for ERA of FFC Residuals Comanaged in Surface Impoundments

Sectors: comanaged wastes from Utility Coal, FBC
Waste Concentrations: coal ash residuals (fly ash, bottom ash, FGD sludge), FBC ash
Waste Management: Surface Impoundment (settling basin)

Strategy: The surface impoundment scenario is constructed similar to the human health risk (e.g., surface area for comanaged coal ash impoundments bounds all surface impoundments). Given the number of studies that were identified on this scenario and the current interest in metals toxicity on amphibians, this scenario may contain the most relevant exposure pathways to evaluate the potential for adverse ecological effects. However, since effluent discharges into swamps may already be covered under other permits, and drainage systems may include the surface impoundment as well as nearby swamps and wetlands, this exposure scenario presents some interesting issues to resolve (particularly with respect to modeled releases). Few studies were identified that provided a thorough characterization of constituent releases from the surface impoundment.

Pathway	Route	Receptors	Key assumptions, issues, and inputs
direct exposure	direct contact ingestion	all receptors mammals, birds, herps	This exposure pathway is unique to the ecological risk assessment component of this analysis. Surface impoundments are used by a variety of waterfowl and settling basin systems and associated receiving waters may contain a variety of flora and fauna (generally species that are tolerant of metals). Consequently, direct exposures to surface impoundment waters and food chain impacts on wildlife that rely on impoundment species as a food source should be considered.
leaching, recharge to aquatic ecosystem, wetlands	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds	The issue of surface water recharge for metals presents a number of complicated, albeit tractable, modeling issues (e.g., speciation in the subsurface; sediment sorption). Although many metals are relatively immobile in the subsurface due to strong binding attractions to organics and charged ions (e.g., clay), metal mobility is strongly influenced by soil and aquifer conditions, including pH and binding capacity (e.g., as a function of anion concentrations such as FeOx). Nevertheless, metal constituents have been shown to seep into wetlands and groundwater from fly ash surface impoundments (Golden, 1983; Guthrie et al., 1982) and amphibians have been shown to be sensitive to the presence of heavy metals associated with coal ash in the ppm to ppb range (e.g., Rowe et al., 1997). Moreover, recent studies have confirmed that the gills are the primary target of metal toxicity (particularly acute toxicity) and, therefore, these effects may be additive. While the ecological significance of such exposures is a difficult question to answer (e.g., is the spatial extent of contamination sufficient to significantly impact wildlife populations and communities), this pathway should be considered in future analyses.

Table 6-4. Development of Exposure Scenario for ERA of FFC Residuals Comanaged in Surface Impoundments

Pathway	Route	Receptors	Key assumptions, issues, and inputs
overland transport to terrestrial ecosystem	direct contact ingestion	soil biota, plants mammals, birds, herps	The same limitations apply to this pathway as with the human health scenario. For example, the overland pathways are believed to be highly unlikely since: (1) surface impoundments are generally not above grade and (2) existing regulatory authority is considered sufficient to mitigate the risks from overflow due to storm events. Although fish kills have been observed following collapse of settling basins (e.g., Carlson and Adriano, 1993; Cherry et al., 1984), catastrophic failures of surface impoundments were not included in this bounding scenario since fault analysis would be required to estimate the probability of failure. Interestingly, a follow up study two years after the spill indicated that while recovery was occurring, ecological impacts were still observed (e.g., reduction in diversity of bottom fauna).
overland transport in watershed to aquatic ecosystem	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds, herps	This pathway has generated considerable interest by researchers, with studies initiated in the 1970's and continuing to the present day. Carlson and Adriano (1993) identified a number of potential impacts of fly ash disposal in settling ponds on adjacent aquatic ecosystems, including water salinization, decreased photosynthesis, reduced animal reproduction, decreased species densities, and bioaccumulation of metals and metalloids, particularly in aquatic plants. Other authors (e.g., Cherry et al., 1984) noted that contamination of aquatic ecosystems by fly ash probably represent a chronic stress to organisms in a swamp drainage system. In contrast, the man-made wetland associated with an old oil ash/scrubber sludge impoundment in Arizona has provided an enormously important wintering habitat for sandhill cranes suggesting, in the least, that site-specific conditions (e.g., soil type, hydrology) may mitigate the potential for toxicity.

Table 6-5. Development of Exposure Scenario for ERA of Comanaged FFC Residuals Applied as a Soil Supplement in Land Application

Sector: FBC wastes
Waste Concentrations: FBC ash
Waste Unit: land application

Strategy: The land application of FBC ash scenario, as constructed for the ecological analysis, is very similar to the landfill scenario. Because it is primarily applied to agricultural lands as a soil amendment, it is expected that the land use patterns for the areas receiving FBC ash contained fragmented terrestrial habitats. Consequently, a limited variety of ecological receptors is likely to be exposed to FBC ash. In addition, Carlson and Adriano (1993) point out that the results of studies conducted up to 1993 indicate that FBC ash can be used as a lime substitute, and as a source of CA and S on problem soils. Relatively few problems have been associated with its use (e.g., high alkalinity and salinity), and metals toxicity has been absent in plant studies. Accumulation of metals of metals at levels of concern in animals has not been shown when animals are fed plants grown on FBC-amended soils (Cochran et al., 1991). In fact, a number of studies demonstrated improved crop yields and overall improvement of soil characteristics. Transport of metal constituents from land-applied FBC ash applications to wetlands and aquatic ecosystems is possible and should be considered.

Pathway	Route	Receptors	Key assumptions, issues, and inputs
overland transport to terrestrial ecosystem	direct contact ingestion	soil biota, plants mammals, birds, herps	Study data suggest that food chain impacts are absent from FBC application for plants and, therefore, exposures to herbivores need not be evaluated (Carlson and Adriano, 1993). Potential exposures to birds and mammals that feed primarily on earthworms and insects should be considered, however, the nature of FBC ash application suggests that this pathway is likely to be of little concern.
overland transport in watershed to aquatic ecosystem	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds	Proximity of agricultural field to aquatic ecosystem will be a key determinant of whether the potential for adverse effects to aquatic biota and associated wildlife exists. Since the toxicity of many metals to aquatic biota and amphibians tends to increase with decreasing pH (e.g., Freda, 1991), metal contamination from FBC ash may be bounded out by the landfill scenario. For example, FBC ash is alkaline and coal fly ash is highly variable and includes fairly acidic wastes, particularly those that contain pyrites. Nevertheless, impacts on sediment communities may be of potential concern.

Table 6-5. Development of Exposure Scenario for ERA of Comanaged FFC Residuals Applied as a Soil Supplement in Land Application

Pathway	Route	Receptors	Key assumptions, issues, and inputs
emissions, dispersion, deposition to terrestrial ecosystem	direct contact ingestion	soil biota, plants mammals, birds, herps	As with the human health scenario, environmental settings favorable to windblown emissions (e.g., high wind speeds) should be used to screen this pathway. Considering the results of studies on phytotoxicity (of food crops) and on animal toxicity, it would seem unlikely that windblown emissions would approach constituent concentrations observed on the actual application site. Hence, exposures from windblown emissions and deposition may be substantially less than on-site exposures for which there are data. As with the landfill scenario, total metal concentrations are used to approximate the mass available for transport and air emission factors may be estimated from AP-42.
emissions, dispersion, deposition to aquatic ecosystem, wetlands	direct contact ingestion	aquatic & sediment biota, amphibians mammals, birds	Depending upon the topography of the modeled site, this exposure pathway may contain the most significant release and transport mechanism of FBC ash to aquatic ecosystems. It seems somewhat unlikely that appreciable mass of FBC would be blown to a nearby surface water body for certain applications of FBC ash (e.g., orchards) and, therefore, the application use should be considered in evaluating potential transport.

6.3 ANALYSIS PHASE

This section covers the methodology used to develop the exposure profiles specific to the scenario/residual combinations in Table 6-2 as well as the methodology used to estimate ecotoxicological benchmarks and chemical stressor concentration limits (CSCLs), as appropriate, for receptors evaluated in this screening analysis. The fate and transport model used to estimate CPEC concentrations in plants, soil, sediment, and surface water (i.e., **exposure profile**) is discussed in previous sections in this report and the derivation of CPEC concentrations for comanaged FFC residuals is described in detail in the SAIC Background Document. Consequently, the discussion relative to the exposure profiles is intentionally brief. The methodology used to support the development of CSCLs for the FFC ERA (e.g., **stressor-response profiles**) is explained in some detail in Sections 6.3.2 and 6.3.3.

6.3.1 Exposure Profiles

As described in Section 4 of this report, the exposure profiles were generated using the fate and transport model under development for the HWIR and currently in use by a number of other projects at the Office of Solid Waste (OSW). Depending on the assessment tier, the model provided medium-specific concentrations of CPECs for bounding, high-end, and central tendency management and use practices.

6.3.2 Stressor-Response Profiles

For each CPEC, a stressor-response profile (i.e., the quantitative relationship between exposure and effect) was created to serve as the basis for ecotoxicological benchmarks and CSCLs. For the purposes of the FFC ERA, benchmarks are defined as effects levels in units of dose (mg/kg-day) that are relevant to population sustainability. Chemical stressor concentration limits (CSCLs) are defined as effects levels in units of concentration (ppm) that are relevant to: (1) population sustainability (i.e., CSCL is based on the benchmark), or (2) community structure and function (i.e., CSCL is based on data on individual species assigned to the community). Ecotoxicological data are collected on acute endpoints (e.g., lethality), chronic endpoints (e.g., growth, reproduction), and on bioaccumulation potential to develop CSCLs that reflect the nature of exposure (i.e., direct or food web).

It is useful to think about the CSCLs developed from stressor-response profiles in terms of either population-type concentration limits or community-type concentration limits. The population-type concentration limits generally reflect exposures via ingestion of contaminated media and food items (e.g., plants, prey). One exception to this tenet is for amphibian populations; the CSCLs were based on direct aqueous exposures over subchronic durations for endpoints generally limited to measures of lethality. The community-type concentration limits generally reflect direct exposures to contaminated media.[8] It should be noted that the CSCLs for receptor communities are not truly *community-level* concentration limits in the sense that they do not consider predator-prey interactions. Rather, they are based on the theory that protection of 95% of the species in the community will provide a sufficient level of protection for the community (see, for example, Stephan et al., 1985, for additional detail).

The CSCLs developed for constituents found in FFC residuals are consistent with the proposed guidelines for ecological risk assessment. For example, point estimates of CSCLs were derived for no observed adverse effects levels (NOAELs may be applied for T&E species or in conservative analyses), lowest observed adverse effects levels (LOAELs), or other *de minimis* effects levels for selected endpoints (e.g., protection of 95% of the species). The overall methodology used to derive benchmarks and CSCLs for the FFC ERA is described briefly below.

6.3.2.1 Benchmark and CSCL Development for Wildlife Populations

For **amphibian** populations, the development of CSCLs was severely limited by data availability. After a review of several compendia presenting amphibian ecotoxicity data (e.g., EPA, 1996b; Power et al., 1989) as well as primary literature sources, it was determined that the lack of standard methods on endpoints, species, and test durations made deriving a chronic CSCL for amphibians inappropriate. Consequently, an acute CSCL was derived for aqueous exposures in amphibians by taking a geometric mean of LC_{50} data identified in studies with exposure durations less than eight days. Although the use of acute effects levels is not consistent with other benchmarks and CSCLs, the sensitivity of these receptors warrants their inclusion even though chronic concentration limits have not yet been developed. Other methods to address chronic exposures to amphibians are currently being evaluated. One alternative under consideration is the addition of amphibian toxicity data to the data set used to the Ambient Water Quality Criteria (see Stephan et al, 1985) since the AWQC methodology allows for amphibians to be included among the representative taxa.

For populations of **mammals and birds**, the overall approach used to establish ecotoxicological benchmarks was similar to the methods used to establish RfDs for humans as described in the Integrated Risk Information System (IRIS). Each method uses a hierarchy for the selection of toxicity data and extrapolates from a test species to the species of interest. However, there are fundamental differences in the goals of noncancer risk assessments for humans and ecological receptors. Risk assessments of humans seek to protect the individual while risk assessments of ecological receptors typically seek to protect populations or communities of important species. The procedures used to develop benchmarks (i.e., RfDs) for the protection of human health are very sensitive by design, and go beyond the need to sustain the reproductive

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[8] Several Ambient Water Quality Criteria (e.g., for DDT) are based on Final Residue Values that reflect the bioaccumulation of contaminants in aquatic organisms. However, most AWQC are based on direct effects to aquatic organisms.

fitness in a local population (U.S. EPA, 1992b). Consequently, benchmarks for mammals and birds were established using three key guidelines.

- First, because population viability was selected as an assessment endpoint, the benchmarks were developed from measures of reproductive/developmental success or, if unavailable, other effects that could conceivably impair population dynamics.

- Second, the population-level benchmark is preferred over population-inference benchmarks (i.e., LOAELs for individual organisms on reproductive endpoints). Although relatively few population-level benchmarks have been developed to date, these benchmarks are considered to be more rigorous than the point estimates gleaned from toxicity studies.

- Third, uncertainty factors (UFs) are generally not applied to address interindividual variability. For example, a UF of 10 was not applied to subchronic studies since reproductive and developmental toxicity studies are frequently short-term.

Once the benchmark study was identified, the CSCL was calculated for each medium of interest using a three step process. The remainder of this section outlines the basic technical approach used to convert avian or mammalian benchmarks (in daily doses) to the CSCL (in units of concentration) for surface water and soil. The methods reflect exposure through the ingestion of contaminated plants, prey, and various media and include parameters on accumulation (e.g., bioconcentration factors), uptake (e.g., consumption rates), and dietary preferences.

STEP 1

Scale benchmark: The benchmarks derived for various taxa (e.g., mammals) can be extrapolated to other species within a taxa by the cross-species scaling equation (Sample et al., 1996). For population-level benchmarks, this extrapolation integrated into the population model framework. For population-inference benchmarks, the extrapolation is performed using Equation.

$$Benchmark_w = LOAEL_t \ x \left(\frac{bw_t}{bw_w} \right)^{1/4} \qquad \text{(Eqn 1)}$$

where $LOAEL_t$ is the LOAEL for the test species, bw_w is the body weight of the wildlife species, and bw_t is the body weight of the test species. This is the default methodology EPA proposed for carcinogenicity assessments and reportable quantity documents for adjusting animal data to an equivalent human dose (57 FR 24152). It should be noted that at least one recent paper demonstrated that cross-species scaling may be problematic for avian species (Mineau et al., 1996).

STEP 2

Identify BCF/ BAFs: For metal constituents, whole-body bioconcentration (BCFs) and bioaccumulation factors (BAFs) were identified or estimated, as appropriate, for aquatic and terrestrial organisms that may be used as food sources (e.g., fish, plants, earthworms). The Oak Ridge National Laboratory has recently proposed methods and data that are useful in predicting bioaccumulation in earthworms and small mammals (Sample et al, 1998a, 1998b). In short, these values are typically identified in the open literature and EPA references, or calculated based on the relationship between log K_{ow} and accumulation in lipid tissue. Currently, distributions of BAFs are under development for HWIR98 that reflect variability across hydrological regions as well as various fish species and aquatic food webs.

STEP 3

Calculate CSCLs: The following equation provides the basis for calculating the CSCL for surface water using a population-inference benchmark (e.g., endpoint on fecundity).

$$Benchmark\ (mg/kg-day) = \frac{[I_{fish} \times (BAF \times C_w^t)] + (I_w \times C_w^t)}{bw} \qquad \text{(Eqn 2)}$$

where:

I_{fish} = intake of contaminated fish (kg/d)
BAF = whole-body bioaccumulation factor (L/kg)
bw = weight of the representative species (kg)
I_w = intake of contaminated water (L/d)
C_w^t = total concentration in the water (mg/L)

and the whole-body concentration in fish (C_{fish}) is given by BAF x C_w^t.

For chemicals that bioaccumulate significantly in fish tissue, the ingestion of contaminated food will tend to dominate the exposure (i.e., [I_{fish} x C_{fish}] >> [I_w x C_w^t]), and the water term (i.e., [I_w x C_w^t]) can be dropped from Eqn 2 resulting in Eqn 3.

$$Benchmark\ (mg/kg-day) = \frac{I_{fish} \times (BAF \times C_w^t)}{bw} \qquad \text{(Eqn 3)}$$

At the benchmark dose (mg/kg-day), the concentration in water is equivalent to the chemical stressor concentration limit for that receptor as a function of body weight, ingestion rate, and the bioaccumulation potential for the CPEC. Hence, Equation 3 can be rewritten to solve for the CSCL in surface water ($CSCL_{sw}$) as follows:

$$CSCL_{sw} = \frac{benchmark \times bw}{I_w + (I_{fish} \times BAF)} \qquad \text{(Eqn 4)}$$

For wildlife populations of mammals and birds in terrestrial systems, the $CSCL_{soil}$ for a given receptor is given by Equation 5:

$$CSCL_{soil} = \frac{benchmark\ x\ bw}{(I_{food} \sum BCF_j\ x\ F_j\ x\ AB_j) + (I_{soil})} \qquad \text{(Eqn 5)}$$

where

bw	=	body weight (kg)
I_{food}	=	total daily food intake of species (kg/d)
I_{soil}	=	total daily soil intake of species (kg/d)
BCF_j	=	bioaccumulation factor in food item j (assumed unitless)
F_j	=	fraction of consisting of food item j (unitless)
AB_j	=	absorption of chemical in the gut from food item j

6.3.2.2 CSCL Development for Communities

For the **aquatic community**, the Final Chronic Value (FCV) developed for: (1) the Great Lakes Water Quality Initiative, or (2) the National Ambient Water Quality Criteria (NAWQC) was the preferred sources for the $CSCL_{sw}$. If an FCV was unavailable and could not be calculated from available data, a Secondary Chronic Value (SCV) was estimated using methods developed for wildlife criteria estimated for the Great Lakes Initiative (e.g., 58 FR 20802). The SCV methodology is based on the original species data set established for the NAWQC, however, it requires fewer data points and includes the statistically derived adjustment factors. For CSCL derivation, the minimum data set required at least one data point for daphnids.

For the **sediment community**, the approach used to establish CSCLs for metals is based on a complete assessment of several sources proposing sediment benchmark values. For nonionic organic chemicals, the methods used were adapted from the EPA guidance: *Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using Equilibrium Partitioning* (U.S. EPA, 1993).

For **algae and aquatic plants**, adverse effects concentrations are identified in the open literature or from a data compilation presented in *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision* (Suter and Tsao, 1996). For most contaminants, studies were not available for aquatic vascular plants and lowest effects concentrations were identified for algae. The CSCL for algae and aquatic plants was based on (1) a lowest observed effects concentration (LOEC) for vascular aquatic plants or (2) an effective concentration (EC_{xx}) for a species of freshwater algae, frequently a species of green algae (e.g., *Selenastrum capricornutum*). Because of the lack of data for this receptor group and the differences between vascular aquatic plants and algae sensitivity, the lowest value of those identified was usually chosen.

For the **terrestrial plant community**, ecotoxicological data were identified from a summary document prepared at the Oak Ridge National Laboratory: *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants: 1997 Revision* (Efroymson et al., 1997a). The measurement endpoints are generally limited to growth and yield parameters because: (1) they are the most common class of response reported in phytotoxicity studies and, therefore, will allow for criterion calculations for a large number of constituents, and (2) they are ecologically significant responses both in terms of plant populations and, by extension, the ability of producers to support higher trophic levels. As presented in Efroymson et al. (1997a), $CSCL_{soil}$ for phytotoxicity are selected by rank ordering the LOEC values and then approximating the 10^{th} percentile. If there are 10 or fewer values for a chemical, the lowest LOEC is used. If there are more than 10 values, the 10^{th} percentile LOEC was used.

Sources Evaluated for Developing Sediment Community CSCLs

Approach to the Assessment of Sediment Quality in Florida Coastal Waters Volume 1- Development and Evaluation of Sediment Quality Assessment Guidelines. Florida Department of Environmental Protection. (FDEP, 1994)

National Oceanic and Atmospheric Administration (NOAA), (Long and Morgan, 1991)

Screening Level Ecological Risk Assessment at Hazardous Waste Combustion Facilities. Office of Solid Waste (U.S. EPA, 1997)

Eco Update. Office of Emergency and Remedial Response. (U.S., EPA, 1996)

Oak Ridge National Laboratory. Screening Level Benchmarks for Ecological Risk Assessment. U.S. Department of Energy. (ORNL, 1996)

Technical Support Document for the Hazardous Waste Identification Rule: Risk Assessment for Human and Ecological Receptors. 1995 (U.S. EPA, 1995)

For the **soil community**, CSCLs are developed using methods analogous to those used in deriving the NAWQC. In brief, the $CSCL_{soil}$ values for soil fauna were estimated to protect 95% of the species found in a typical soil community, including earthworms, insects, and other various soil fauna. Microflora have not been included in the soil community primarily because of the difficulty in assigning ecological significance to effects levels for soil microorganisms. However, appropriate methods to add microflora are currently under evaluation since: (1) microflora make up approximately 80-90% of the biomass in soil and (2) microflora are responsible for the majority of the biological activity in soil (e.g., N mineralization).

In developing CSCLs, eight taxa of soil fauna were identified to capture the key structural (e.g., trophic elements) and functional (e.g., decomposers) components of the soil ecosystem. The methodology presumes that protecting 95% of the soil species with a 50^{th} percentile level of confidence will ensure long-term sustainability of a functioning soil community. The toxicity data on soil fauna are gleaned from several major compendia and supplemented with additional studies identified in the open literature. Generally, the studies are not evaluated in terms of quality because there is currently no consensus on standard methods and species for soil testing (although earthworms are frequently used as indicator species). However, acceptable toxicity data are limited to soil studies (vs aqueous studies) on measurement endpoints believed to be relevant to population survival (e.g., growth, reproduction).

The approach to calculating benchmarks for the soil community is based on efforts by Dutch scientists (i.e., the RIVM methodology) to develop hazardous concentrations (HC) at specified levels of protection (primarily 95%) at both a 95th percentile and a 50th percentile level of confidence (Sloof, 1992). For the soil fauna benchmarks, the 50th percentile level of confidence was selected because the 95th percentile appeared to be overly conservative for a "no effects" approach. The RIVM methodology follows two steps: (1) fitting a distribution to the log of the selected endpoints, and (2) extrapolating to a benchmark concentration based on the mean and standard deviation of a set of endpoints. The key assumptions in the Dutch methodology are that: (1) LOEC data are distributed logistically, and (2) the 95% level of protection is ecologically significant. The following formula was used to calculate soil fauna benchmarks:

$$HC_{5\%} = [x_m - k_l \ s_m] \qquad \text{Eqn 6}$$

where

$HC_{5\%}$	=soil concentration protecting 95% of the soil species
x_m	=sample mean of the log LOEC data
k_l	=extrapolation constant for calculating the one-side leftmost confidence limit for a 95% protection level
s_m	=sample standard deviation of the log LOEC data

It is important to note that only one value for k_l is calculated for the 50th and 95th percentile confidence limits, respectively, for each sample size (m). Consequently, it is assumed that: (1) there is just one extrapolation constant with the required confidence property for each species sample size, and (2) extrapolation factors may be determined through Monte Carlo simulation by generating random sample averages and deviations for the *standard* logistic distribution and adjusting for a specified confidence level (i.e., 50th or 95^{th}). When data are insufficient for CSCL development, benchmark studies identifying effects to earthworms and other soil biota proposed by Oak Ridge National Laboratory (Efroymson et al., 1997b) or criteria developed by the Canadian Council of Ministers of the Environment (CCME, 1997) were used to estimate protective soil concentrations.

6.4 Risk Characterization

EPA defines the risk characterization in terms of: (1) the risk estimation which integrates the exposure and stressor-response profile to estimate the likelihood of adverse ecological effects and (2) the risk description which synthesizes the overall conclusion of the assessment and addresses uncertainty, assumptions, and limitations. For assessments that are based upon hazard quotient (HQ) approach,[9] the comparison of modeled exposure concentrations to CSCLs in the risk estimation has a binary outcome: either the constituent concentration is above the screening criteria (HQ>1) or the concentration is below the criteria (HQ<1). Since the CSCLs are based on *de minimis* ecological effects, it is presumed that a hazard quotient below 1 indicates a low potential for adverse ecological effects for those receptors included in the analysis for which data are available. However, it is important to recognize that the HQ methodology is designed to delineate the **potential** for adverse ecological effects as a function of HQ exceedances. Although this method provides important insight into the potential for adverse ecological effects, the results are relevant only to those receptors that were included in the assessment and for which data were available. The results have limited utility in interpreting the ecological significance of predicted effects and caution should be exercised in extrapolating to ecosystems (e.g., wetlands) and receptors (e.g., threatened and endangered species) not explicitly modeled in this framework.

6.4.1 Risk Estimation

This section presents the risk results (i.e., hazard quotient exceedances) from the ERA for bounding, high-end, and central-tendency management and use practices for comanaged FFC residuals. The risk results from the bounding assessment were used to define the list of CPECs evaluated for each sector and management/use practice for the high-end and central tendency analyses.[10] Table 6-6 presents the constituents considered for each of the eight sector and management/use combinations considered in the ERA.

The hazard quotient results for the high-end and central tendency surface impoundments are presented in Table 6-7. Although no HQ exceedances were observed for any off-site exposures (i.e., receptors not located on the WMU), exceedances were identified for direct use of the surface impoundment by receptors associated with aquatic systems for bounding, high-end, and central tendency management and use practices. For the assessment of surface impoundment use by ecological receptors, the "management and use practices" are defined simply as the maximum, high-end (i.e., 95^{th} %ile), and central tendency (i.e., median value) CPEC concentrations in the comanaged, coal ash surface impoundment waters. For the high-end assessment, HQ exceedances indicated the potential for acute effects in amphibians (direct exposure) and chronic effects in mammals and birds associated with aquatic ecosystems (direct &

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9 In the jargon of ecological risk assessment, the HQ approach is often referred to as a screening approach.

10 In almost all cases, the high-end parameters were concentration and surface area of the WMU.

food chain exposure). For the central-tendency assessment, HQ exceedances were only observed for mammals and birds exposed to lead, mercury, and selenium. All HQ results are presented in Appendix J.

Table 6-6. CPECs Included in the ERA of High-end and Central-tendency Management and Use Practices (by sector and waste management unit)

Management and Use Scenarios	Constituents
Utility Coal Co-managed Coal Waste	
Landfill	Barium, Lead, Cadmium, Selenium, Silver, Aluminum, Boron, Cobalt, Thallium
On-site Surface Impoundment*	Aluminum, Antimony, Arsenic, Barium, Boron, Cadmium, Chromium, Cobalt, Copper, Lead, Mercury, Molybdenum, Nickel, Selenium, Silver, Thallium, Vanadium, Zinc
Dewatered Surface Impoundment	Barium, Lead, Cadmium, Selenium, Silver, Aluminum, Boron, Cobalt, Thallium
Fluidized Bed Combustion (FBC) Combined Ash Wastes	
Landfill	Cadmium, Lead, Nickel, Vanadium, Silver, Aluminum, Boron, Cobalt, Thallium
Agricultural Soil Amendment	Nickel, Vanadium, Aluminum, Boron, Cobalt, Thallium
Utility Oil-Fired Wastes (Solids Settling Basin Ash)	
Landfill Onsite	Arsenic , Cadmium, Copper, Lead, Nickel, Vanadium, Zinc, Chromium, Silver, Aluminum, Boron, Cobalt, Thallium
Non-Utility Coal Co-Managed Wastes	
Landfill Onsite	Barium, Lead, Cadmium, Selenium, Silver, Aluminum, Boron, Cobalt, Thallium
Landfill Offsite	Barium, Lead, Cadmium, Selenium, Silver, Aluminum, Boron, Cobalt, Thallium
* Measured concentrations identified from EPRI site investigations.	

6.4.2 Risk Description

The risk description attempts to integrate all of the qualitative and quantitative information developed during the screening ERA and to interpret the results for use in the decision-making process. The lines of evidence are examined based on the relevance of the assessment endpoints, the sufficiency of the data presented in the exposure and stressor-response profiles, the appropriateness of the methods used to predict ecological effects. Interpreting the impact of an HQ exceedances requires an understanding of both the derivation (data and methods) of CSCLs as well as insight into the CPEC concentrations predicted by fate and transport models.

Ecological risk characterizations, especially predictive ones such as the FFC ERA, often require the development of assumptions to characterize emissions, sites, receptors, exposure, and spatial and temporal factors; the risk results must be interpreted within the context of the limitations inherent in a generalized approach (versus a *site-based* analysis). Although tools to evaluate HQ exceedances within the context of ecosystem function and structure are not available for HQ-based approaches, key assumptions and uncertainties should be examined in delineating ecological significance. The major limitations and uncertainties in the FFC ERA can

Table 6-7. Ecological Risk Assessment Results for Surface Impoundments

	High-end Assessment	**Central-tendency Assessment**	**Receptor Group**
Aluminum	10	2	amphibians
Antimony	--	--	no exceedances
Arsenic	19	--	birds
Barium	--	--	no exceedances
Beryllium	--	--	no exceedances
Boron	16	--	amphibians
Cadmium	23	--	mammals
Chromium	--	--	no exceedances
Cobalt	--	--	no exceedances
Copper	--	--	no exceedances
Lead	830	45	mammals
Mercury	4,500	3,100	birds
Molybdenum	--	--	no exceedances
Nickel	--	--	no exceedances
Selenium	30,000 & 5	150	mammals & amphibians
Silver	--	--	no exceedances
Thallium	--	--	no exceedances
Vanadium	--	--	no exceedances
Zinc	--	--	no exceedances

be grouped under: (1) exposure issues, (2) CSCL development, and (3) receptors/ecosystems at risk. While certain limitations are intrinsic to *any* ecological risk assessment (e.g., extrapolation of laboratory data to field exposures; application of bioaccumulation factor ratios), this section is focused on the uncertainty, limitations, and the confidence specific to the ERA of comanaged FFC residuals.

6.4.2.1 Exposure Issues.

Co-occurrence of Receptor and CPEC - As a simplification for national-scale analyses (i.e., no site-based data), co-occurrence is typically assumed. However, the prior probability that a receptor will be found in a contaminated sector is not known nor is it known whether a receptor will forage for food in contaminated areas or if those areas do, in fact, support the type of habitat needed by the receptor. Although the assumption of co-occurrence was necessary for the FFC ERA, the increase in resolution offered by GIS tools and other data sources greatly improves the strength of that assumption (e.g., topo maps; local FWS agencies). Nevertheless, the co-occurrence of the stressor and the assessment endpoint must be demonstrated to the extent supported by the data.

Assumptions on Dietary Exposure - National-scale assessments often assume maximum intake of contaminated prey in the diets of primary and secondary consumers (i.e., 100 percent of the diet originates from the contaminated area). Obviously, under field conditions many receptors are opportunistic feeders with substantial variability in both the type of food items consumed as well as the seasonal patterns of feeding and foraging. Consequently, the exclusive diet of contaminated food items tends to provide a very conservative estimate of potential risks. With the introduction of site-based data on wildlife use of surface impoundments, it is possible to adjust dietary exposure estimates to reflect the food web structure of nearby habitats.

Bioavailability of Constituents of Concern - For the purposes of this analysis, all forms of a constituent are assumed to be equally bioavailable and, therefore, the actual exposures that may occur in the field tend to be overestimated. This assumption is appropriate for a conservative analysis; however, both the chemical form and the environmental conditions influence bioavailability and, ultimately, the expression of adverse effects. For example, the form of arsenic has been shown to profoundly influence mobility and toxicity. The stressor-response profiles developed for metal CPECs should include at least a qualitative discussion of the speciation behavior of metals in various environmental media.

Multiple Constituent Exposures - The risk of each CPEC was considered separately in this analysis. However, the waste concentration data on FFC residuals comanaged in surface impoundments suggest that exposure to multiple constituents is highly likely. The synergism or antagonism between different constituent combinations may elicit unexpected adverse impacts to ecosystems. Hence, a single-CPEC analysis may underestimate risks associated with multiple chemical stressors.

CPEC Concentrations in FFC Residuals - One of the most significant sources of uncertainty in the ecological risk analysis is the CPEC concentrations selected for the surface impoundment scenario. As described in Appendix B (Concentration Data Used for FFC Risk Assessment - Table B-5), the sample population for CPECs is relatively small. In the case of mercury, for example, only two samples were available and one of the samples was below the detection limit. As a result, it is difficult to determine the potential significance of HQ exceedances shown in this study with respect to entire universe of comanaged FFC residuals.

6.4.2.2 CSCL Development

Data Gaps - CSCLs were developed for constituents when sufficient data are available. In many cases, sufficient data were unavailable for a receptor/constituent combination and, therefore, the potential risk to a receptor can not be assessed. In particular, insufficient data were available to derive chronic effects CSCLs for amphibians. Because the risk results can only be interpreted within the context of available data, the absence of data can not be construed to mean that adverse ecological effects will not occur.

Conservatism of CSCL Development - Because the overall approach is based on “no effects” or “lowest effects” study data, these criteria tend to be fairly conservative. In site-specific assessments, a *de minimis* effects approach is often replaced with an effects level similar to natural population variability (e.g., sometimes as high as a 20% effects level). As a result, the CSCLs may be overly conservative for representative species and communities assigned to the habitats of interest. Since the difference between a LOAEL and a NOAEL is often about a factor of 10, an HQ exceedances around 10 may not be ecologically significant (e.g., high-end arsenic exposure in birds). In contrast, CSCLs based on no effects data that are developed for the protection of T&E species are presumed to be appropriately conservative.

6.4.2.3 Receptors/Ecosystems at Risk

One of the most intractable problems in conducting a predictive ERA intended to reflect risks at a national scale is evaluating *all* of the receptors and ecosystems at risk. In the *Wastes from the Combustion of Coal by Electric Utility Power Plants - Report to Congress* (EPA, 1988), the authors pointed out that plants or animals of concern were located within a 5 km radius of the FFC waste management units at between 12% and 32% of the sites. Although these figures are based on the Biological and Conservation Data (BCD) System and are, therefore, of limited spatial resolution, they suggest the possibility that threatened and endangered species and/or critical habitats may be at risk from comanaged FFC residuals. In particular, the impacts of comanaged FFC residual releases on estuarine systems and associated receptors (e.g., manatees) has been raised internally by OSW staff. Examples of other critical assessment endpoints not evaluated in the FFC ERA include the following.

Managed lands - Because ecosystem degradation is proceeding at an unprecedented rate, and because protected lands play a critical role in preserving plant and animal species, managed areas in the United States represent well-recognized ecological values. Managed lands refer to a variety of lands designated by the federal government as worthy of protection, including National Wildlife Refuges, National Forests, Wilderness areas, and National Recreation areas.

Critical habitats - Although critical habitats may be defined in a number of ways (e.g., presence of threatened species; decreasing habitat area), wetlands are widely recognized as serving critical ecological functions (e.g., maintenance of water quality). The U.S. Fish and Wildlife Service estimates that approximately 45% of the Nation's threatened and endangered species directly depend on aquatic and wetland habitats. Consequently, impacts of chemical stressors on wetland habitats may have high ecological (and societal) significance. The presence of critical habitats such as wetlands is also used to inform the selection of ecological receptors (e.g., amphibians; waterfowl) and the construction of appropriate food webs.

Threatened and endangered species - For most ecological risk assessments of chemical stressors, available data on toxicity and biological uptake are sufficient to support the evaluation of effects on representative species populations or generalized communities (e.g., aquatic community). However, despite their obvious value, threatened and endangered (T&E) species, species of special concern, or species in need of conservation are frequently excluded from the analytical framework for national rulemakings. The assessment of T&E species requires a site-based approach in which locations, habitats, and species of concern are identified and characterized with respect to the spatial scale of CPEC releases. A number of sources of information on T&E species are available, ranging from a county-level database developed by the EPA Office of Pesticide Programs that lists T&E species, to a network of state and local contacts responsible for the maintenance and administration of the National Biological Database. In addition, there are other federal agencies such as the Department of Defense that are in the process of developing wildlife species profiles for T&E species.

7.0 UNCERTAINTY

Previous sections of this report have presented the principal data, assumptions, and models used to develop risk results for FFC wastes. This section qualitatively addresses the primary sources of uncertainty within the risk assessment and the effects that this uncertainty has on interpreting the results. The discussion is intended to provide a basis for developing conclusions regarding the risk results. It is important to note that the uncertainties discussed below are relevant to both the human health and ecological risk assessment. For both receptor types, the major uncertainties are associated with the characterization of FFC wastes and waste management practices. Key uncertainties that are exclusive to the ecological risk analysis are described in Section 6.4 - Risk Characterization.

Uncertainty in an analysis such as this can be introduced in a number of places. In general, the major sources of uncertainty are parameter uncertainty and model uncertainty. Parameter uncertainty occurs when parameters appearing in equations cannot be measured precisely and/or accurately. Model uncertainty is associated with all models used in all phases of a risk assessment. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in models due either to increased complexity or to a lack of data on that variable.

Most of the issues addressed in this section are specific to this project. These issues include uncertainties regarding waste characterization, waste management practices, and uncertainties introduced with the models that were used. The exception to the analysis specific uncertainty discussion will be a brief discussion on chemical specific uncertainty issues; specifically, metal speciation in the environment. Each of these issues will be discussed below. The discussion on uncertainty will discuss the possibilities for a follow on, quantitative uncertainty analysis.

Waste Characterization

Waste characterization is the main uncertainty issue for this study. Review of Section 5.6 strongly suggests that waste concentration is the single most important parameter in determining human health risk. The sensitivity of the model to this parameter, the small sample size for the waste characteristics data, and the high demonstrated variability of waste characteristics across facilities presents significant uncertainty issues for EPA.

Sample size is an important issue for all waste streams. However, sample size is particularly small relative to the size of the industry for the coal-fired co-managed wastes. The waste characterization is based on 14 Electric Power Research Institute (EPRI) Site Investigations plus two additional EPRI reports. How well this limited sample characterizes the population of coal-fired co-managed wastes is questionable.

For all wastes there were little to no data provided on the physical characteristics, such as particle size distribution. Particle size can have significant impacts on the waste's mobility in the environment. Many of these wastes also have self cementing properties which would greatly retard constituent mobility in the environment. These properties were not characterized in any quantitative assessment that could be incorporated into the analysis. (Even if data had been supplied on these properties, there would be significant model uncertainty in measuring transport from the source.)

Waste characterization uncertainty may be especially important in the dewatered surface impoundment and the oil ash landfill scenarios. In each case, the scenario assumes that exposure occurs after the wastes have been subjected to leaching in an impoundment. Since many constituents of concern may be readily leached from particle surfaces in impoundments, the total concentration of these constituents at the end of the active life of the impoundment (or at the time of transfer to a landfill) may be expected to be lower than at the time of generation. The data provided for these wastes, however, reflect an uncertain extent of chemical and physical alteration resulting from management. Therefore, concentrations used for these scenarios may overstate the true concentration of the waste after such leaching has taken place.

Non-utility combustion processes and the types of fuels used are many and varied. Much uncertainty is introduced into the assessment by assuming that the waste is identical to the utility coal-fired co-managed wastes.

Finally, for all waste sectors, different types of wastes were analyzed (e.g., fly ash, bottom ash, solids settling basin sludge). Decisions were made on how to aggregate or dis-aggregate wastes based on various criteria. These decisions can have significant impact on risk results as this determination will necessarily dictate the waste concentrations, constituents of concern, and other physical characteristics of the waste.

These are some of the primary uncertainty issues for the human health, non-groundwater pathway risk assessment. Other waste characterization uncertainty issues are addressed in Section 7 of SAIC's Background Document.

Waste Management Practices

Many waste management practices have significant impacts on risk results and also have a great amount of uncertainty. Some of the major issues include: landfill cover practices; the use of erosion and runoff controls; waste hauling practices; the actual unit sizes; and actual landfill active lifetime.

There are many options for applying a cover to a landfill. Some options may include: applying no cover; applying a daily cover; disposing of waste in cells and when a cell is filled applying a permanent cap. These choices have significant impacts on the types of emissions that can be expected from a unit.

Many landfills may be required to have runoff and erosion controls installed either due to other environmental regulations or due to permit requirements. There were no information on the prevalence of runoff and erosion controls for the landfills assessed for this analysis.

Waste hauling and disposal practices have significant impacts on particulate air emissions. If dump trucks are used, input parameters that are important in the determining particulate emissions include: capacity of the dump truck; number of tires on the dump truck; and waste cover efficiency, length of road. It is assumed in this analysis that dump trucks were used to haul the waste to the landfill. If other delivery mechanisms are used this could have major impacts on amount of air emissions.

Waste management unit size and active lifetime are also important risk drivers. As with waste characterization, it is questionable whether the small sample size of units accurately represents the universe of disposal units in the nation. This is especially true for the utility oil-fired wastes and the non-utility coal-fired wastes. Because there were not sufficient data to characterize WMU sizes, dimensions were developed based on fuel use. There are many assumptions that are made when going from fuel use to WMU size. Each of these assumptions introduces uncertainty into the analysis.

Model Uncertainties

Assessing model uncertainties is a complex issue and beyond the scope of this report; however, there are a few issues that warrant mentioning. These issues will be mentioned in the conclusion section below when they have the potential to have a significant impact on the risk results that have been calculated.

Metal Speciation

Analytical data provided to RTI were in the form of total metal concentrations, which do not account for metal speciation. As a consequence, the risk analysis did not account for metal speciation in the various environmental compartments. Metal speciation is an important factor in assessing the fate, mobility, and toxicity of metals in the environment. Without a clear understanding of metal speciation, behavior is uncertain. For purposes of the risk assessment, the most toxic metal species were assumed to equal to total metal concentration (e.g., chromium VI was assumed to be equal to total chromium concentration). Although this is considered a conservative assumption, it is important to recognize the uncertainty that this assumption introduces into the analysis.

Quantitative Uncertainty Analysis

Given the level of uncertainty for many of the data for this analysis a quantitative uncertainty analysis should be performed. EPA guidance for performing a probabilistic risk analysis as a quantitative uncertainty/variability analysis states that it is necessary to conduct a sensitivity analysis to identify the risk driving parameters as an initial step. A sensitivity analysis will be will be performed for each constituent, waste, waste management unit, and receptor combination. Once the risk driving parameters are identified a probabilistic (Monte Carlo) analysis may be performed for these wastes.

8.0 CONCLUSIONS

In this study to assess human health and ecological risks from the management and use of the remaining FFC wastes, EPA found that the remaining waste universe is represented by a large and diverse population of wastes and WMU's located throughout the nation. To determine the risks from this large and diverse population, EPA segregated the wastes into four categories: coal-fired utility co-managed wastes, oil-fired utility wastes, FBC wastes, and non-utility coal-fired wastes. Attention was focused on the WMU that EPA believed to present the greatest potential for release of contaminants to the environment. For the non-groundwater pathways, these WMU's included landfills, a dewatered surface impoundment, use of the waste as agricultural soil amendment, and an active surface impoundment (for ecological risks only).

Initially, risks were estimated using a bounding approach where all input parameters are set to their high-end values. The bounding analysis is designed to be conservative; however, this bounding analysis was made even more conservative because the largest waste management unit size that was found for any waste stream/WMU combination was used for all WMU/waste stream analyses (excluding the agricultural soil amendment scenario). The largest WMU size was the dewatered surface impoundment for the utility coal-fired co-managed waste scenario. The size of this unit was 412 acres. This unit size was then used for all landfill scenarios for all waste streams even though no landfill was actually estimated to be that large. Note that for all landfill scenarios except the non-utility offsite commercial landfill, all waste in the landfill is assumed to be the particular FFC waste under consideration. Therefore, larger landfills will necessarily be more conservative as there will be no dilution of the FFC wastes by mixing with other wastes. No waste streams bounded out completely and therefore a more realistic, high-end assessment was undertaken using a deterministic modeling approach.

The following subsections present the major issues to consider in reaching a conclusion for each of the waste categories, a preliminary conclusion for each waste category, and finally presentation of a brief overall conclusion. For each subsection, any exceedences will be discussed with particular attention to the maximum exceedence[11]. Driving input parameters that lead to these exceedences will then be discussed. In addition, results that are close to an exceedence (defined as one order of magnitude) will be discussed.

[11] Because of the recent removal of the ingestion CSF for beryllium, any beryllium exceedences will not be discussed in this section.

8.1 Coal-fired Utility Co-managed Wastes

8.1.1 Coal-fired Utility Co-managed Wastes Managed in Onsite Landfill

Ingestion

Arsenic, barium, and thallium, showed exceedences for ingestion from the landfill scenario. All risks and HQ's are fairly low with the exception of child ingestion risks from arsenic.

For modeling purposes the entire landfill was made up of the coal co-managed wastes and thus starting concentrations were those totals data as presented in Appendix B. The representativeness of this data for the universe of coal-fired co-managed wastes is questionable. It is possible that true waste population concentrations differ from the data used. This could substantially increase or decrease risks depending on the direction degree of the bias.

Conservative assumptions were made regarding the physical characteristics of the waste. For fate and transport modeling, the waste was characterized as an ash. It is likely that the co-managed waste would not retain the properties of the ash and therefore its mobility would be less than that of an ash. Further, coal ash demonstrates self cementing properties. These properties would also retard mobility in the environment; however, the effects of this behavior was not able to be modeled.

It is uncertain whether there are erosion and runoff controls from these landfills. The landfills were modeled under the assumption that there are no controls. If, in fact, there are erosion and runoff controls, exposure and risk estimates will be overstated.

Many of the conservative assumptions that were made for this analysis (e.g., physical properties of the waste, no runoff/erosion controls) would lead to overestimates of risk. Some modeling assumptions would also overestimate risk (e.g., inability to take into account self cementing properties) while other modeling assumptions may slightly underestimate risks (assuming a flat, even-grade source for dispersion modeling). In general, it is believed that the conservative assumptions used in the fate and transport modeling would overestimate risks. However, the driving risk parameter is waste concentration. The waste characterization data also has a great deal of uncertainty. Any bias in the waste characterization data could have significant impacts on the level of risk. A fairly small bias could dictate whether arsenic, thallium, and/or barium exceed risk levels of concern.

Inhalation

Exceedences were only seen for chromium VI at fairly low levels for this pathway. Many of the conservative assumptions regarding the physical characteristics of the waste and the waste management practices would tend to overestimate emissions. Also, total chromium concentrations were modeled as chromium VI. It is highly unlikely that all of the chromium in the waste would be present solely as chromium VI. If the various species of chromium were accounted for individually, the risks from chromium would probably drop.

8.1.2 Coal-fired Utility Co-managed Wastes Managed in Dewatered Surface Impoundment

Ingestion and Inhalation

Arsenic showed exceedences at relatively low risk levels for ingestion and chromium VI showed exceedences at relatively low levels for chromium. The only release mechanism modeled for this scenario were air emissions. As mentioned above, the waste is characterized as an ash. Also, emissions are calculated, and dispersion is modeled, as if the WMU were a flat, even grade unit. In reality, if a surface impoundment were left without official closure, the design characteristics would dictate that the waste would actually be below-grade. This would significantly reduce windblown emissions and dispersion. Also, if the unit were left without official closing, the waste would not retain ash-like properties. It is more likely that the waste would cake over thereby reducing emissions potential. Further, it is documented that often these types of units exhibit opportunistic re-vegetation given sufficient time. This too would retard both emissions and dispersion of particles. Based on these arguments, the risks from this scenario are probably overstated.

8.2 Oil-fired Utility Wastes

The only exceedence observed for this waste stream was for ingestion of arsenic for the onsite landfill scenario. (Note that this is the only waste management unit assessed.)

As with the coal-fired utility co-managed waste, this waste was modeled using fly ash characteristics and assuming year round disposal and emissions. In reality, it is more likely that the surface impoundments, from which this waste is derived, are dredged infrequently (once or twice per year) and these wastes then sent to the landfill. The waste material would be wet, thereby reducing air emissions. It is also likely that the wastes would be covered when disposed of due to the infrequency of the landfill disposal. If not covered, the waste would probably crust over as it dried thereby reducing the potential for emissions. Due to the conservative assumptions of the waste management practices, and fate and transport modeling, the risks from this waste may be overestimated.

8.3 FBC Wastes

Exceedences were observed for ingestion of arsenic and inhalation of chromium VI for the onsite landfill scenario. No exceedences were observed for the agricultural soil amendment scenario.

Ingestion

All of the uncertainty issues discussed in the coal-fired co-managed waste landfill are relevant to this scenario also. Of particular importance to this waste, however, is the issue of the wastes self cementing properties. FBC waste's self cementing properties are more pronounced than those of coal ash. This property would retard migration of waste contaminants and, thus, risks may be over stated.

Inhalation

It is believed that risks from inhalation of chromium are overstated due to the speciation issue mentioned above and due to the fact that the waste's self cementing properties were not able to be incorporated in fate and transport modeling.

8.4 Coal-fired Non-utility Wastes

The only exceedences observed for this waste stream are from ingestion of arsenic. This was observed in both the onsite and offsite landfill. Because the same waste characterization data that were used for the coal-fired utility wastes were used for this waste, all of the issues discussed for ingestion in Section 7.1.1 are relevant here.

8.5 General Conclusions from Human Health Risk Assessment

All risk exceedences for all waste streams were below 1E-5 with the exception of the utility coal-fired co-managed waste/child receptor scenario which had a maximum risk of 1.7E-5. For ingestion pathways, arsenic is the only carcinogen that had exceedences and for the inhalation pathway chromium VI is the only carcinogen that had exceedences. The only non-carcinogens that had an exceedence were barium and thallium with maxium HQ's of 1.0 and 1.1 for the utility coal-fired co-managed waste/child receptor scenario via the ingestion pathway. There were no non-carcinogen HQ exceedences for the inhalation pathway. The general conclusion is that the constituents of most concern are arsenic for ingestion and chromium VI for inhalation.

The driving risk parameter is starting waste concentration. Other important parameters include: area of the WMU, exposure duration, and distance to receptor.

There are several uncertainty issues for this analysis. Most of the assumptions made are conservative and would err to be protective. These assumptions thus may overstate risks. However, the major uncertainty issue for this study is the waste characterization. This is especially important since the driving risk parameter is waste concentration. Any bias in the sampling data that were used to develop the waste characterization would have important effects on the risk results and decisions drawn from this analysis.

8.6 General Conclusions from Ecological Risk Assessment

The risk estimates (i.e., HQ results) for landfills and land application units suggest that ecological risks associated with the release and surface transport of CPECs are not likely to be significant for these management/use practices. Because these results are generally based on no effects levels for mammals and birds, it is expected that even threatened and endangered mammalian and avian species are unlikely to receive exposures that would warrant concern. However, it is difficult to provide unequivocal support to that assertion without the benefit of a more site-based analytical framework. In addition, the subsurface pathway was not evaluated in the FFC ERA, and there is some concern that this pathway may be significant in areas with high water tables that intersect critical wetlands and estuarine systems.

The risk estimates for on-site use of comanaged coal ash surface impoundments indicate that this scenario is of special concern. The potential for adverse effects associated with exposure to CPECs in surface impoundments is support by SAB comments on the HWIR analysis as well as a review of case studies on surface impoundments used to manage FFC residuals. Wildlife (i.e., mammals and birds) and amphibians frequently utilize surface impoundments and nearby wetlands as part of their habitat. Amphibian sensitivity to these trace metals has been confirmed by review of case studies indicating adverse effects to amphibians exposed to FFC coal residuals. The exceedances noted for amphibians indicate the potential for acute effects (i.e., lethality) to amphibian populations inhabiting surface water impoundments. In contrast, the exceedances noted for wildlife indicate the potential for adverse effects to the reproductive capacity of wildlife species. The probability of wildlife being directly exposed to surface impoundment water will vary depending on the surrounding habitat. The most likely receptor to be exposed would be avian receptors that are able to circumvent barriers to forage or nest in pond areas. The lack of prey for avian species (i.e., absence of fish and aquatic invertebrates) may limit the capacity of the pond to support large populations of birds. Although this exposure pathway is only likely to affect a small portion of receptors species, it may have acute impacts to receptors inhabiting these areas.

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